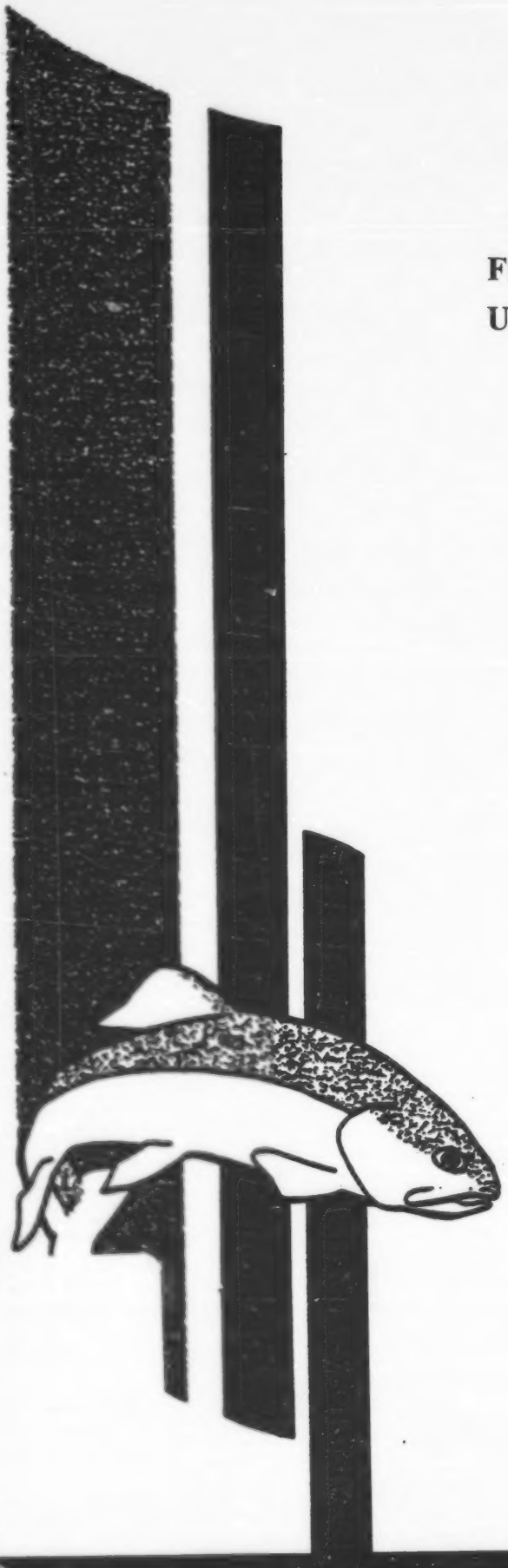

# Sensitivity of British Columbia's Freshwater Fish to Timber Harvest: Using Species Traits as Predictors of Species Risk.

by

Marc Porter
Gord Haas
Eric Parkinson

Fisheries Management Report No. 114
2000

**Province of British Columbia**
BC Fisheries

LEGISLATIVE LIBRARY
VICTORIA BC V8V 1X4 FEB 20 2001

101-02960/

LEGISLATIVE LIBRARY OF BRITISH COLUMBIA
3 3298 00244 5956

# Sensitivity of British Columbia's Freshwater Fish to Timber Harvest: Using Species Traits as Predictors of Species Risk

by

Marc Porter, Gordon Haas and Eric Parkinson

BC Fisheries, Fisheries Research Section
2204 Main Mall, University of British Columbia, Vancouver, BC V6T 1Z4

BC Fisheries
Ministry of Agriculture, Food and Fisheries

Fisheries Management Report No. 114

2000

# ABSTRACT

**Porter, M.S., G.R. Haas and E.A. Parkinson. Sensitivity of British Columbia's Freshwater Fish to Timber Harvest: Using Species Traits as Predictors of Species Risk. BC Fisheries. BC Ministry of Food, Agriculture and Fish. Fisheries Management Report No. 114.**

Assessment of fish response to perturbation can be based upon a biologically integrated, mechanistic framework that examines the functional relationship between species and their environments. The overall effect of timber harvest on fish populations may then be predicted based on relative species tolerances to a range of potentially changing habitat conditions. We evaluated seven major changes to stream and lake habitats associated with logging and identified linkages to twelve associated functional traits displayed by 91 species or subspecies of fish found in British Columbia's freshwaters. Fish taxa were then scored for trait sensitivities on the basis of information from the literature or expert opinion. Overall assessment of relative sensitivity was based on a composite total score for traits and a *k*-means clustering analysis that separated fish taxa into three groups defined as low, moderate or high sensitivity to logging impacts. Our clustering method suggested that approximately 76% of BC's freshwater fish taxa (69 of 91) could be considered to be of moderate or high sensitivity to logging impacts and would suggest that about 29% (26 taxa) could be considered highly sensitive. Fish species that have been introduced and now established in BC waters appear relatively resistant to logging impacts, as most (11 of 15 taxa) scored for low sensitivity. Our scoring of traits provides only a relative metric of sensitivity, however, and full validation of our results is dependent on continuing research into long-term effects of logging impacts on different fish populations. Given the absence of such information for most fish species in the province, functional trait information combined with knowledge of environmental changes may provide the best available method for predicting community response to logging impacts. Fish sensitivity ratings can be combined with information on species distributions (i.e., a rarity score) and associated land use patterns to evaluate current and ongoing risks to individual species. Continued work will focus on quantifying these risk factors using GIS templates, environmental statistics existing for the BC Watershed Atlas, and fish species distribution models derived from historical survey data.

# ACKNOWLEDGEMENTS

This project was conducted with funding from Forest Renewal BC. Our appreciation to Jordan Rosenfeld for his support of the approach and constructive reviews. Juanita Ptolemy, Colin Spence and Bruce Rieman provided helpful comments on an earlier draft of this report.

# TABLE OF CONTENTS


ABSTRACT ..... 1

INTRODUCTION ..... 1

RATIONALE AND METHODS ..... 2

FORESTRY IMPACTS ON STREAM HABITAT ..... 2

*Sedimentation* ..... 3

*Temperature* ..... 4

*Streamflow Patterns* ..... 4

*Riparian Zone* ..... 6

*Large Woody Debris (LWD)* ..... 7

*Stream Productivity* ..... 8

*Fish Passage* ..... 8

CHOICE OF FUNCTIONAL SPECIES TRAITS ..... 9

*Hyperthermia tolerance* ..... 9

*Hypoxia tolerance* ..... 9

*Habitat use* ..... 11

*Siltation tolerance* ..... 11

*Reproductive strategy* ..... 11

*Feeding mode* ..... 12

*Spawning mode* ..... 12

*Migration strategy* ..... 13

*Water column/substrate use* ..... 13

*Spawning season* ..... 13

*Cover use* ..... 13

DERIVATION OF SPECIES TRAIT INFORMATION ..... 14

SCORING OF SPECIES TRAITS USING SENSITIVITY CATEGORIES ..... 15

RESULTS ..... 15

DISCUSSION ..... 19

REFERENCES ..... 23

APPENDIX 1. SCORING FOR FISH SPECIES LIFE HISTORY TRAITS WITH ASSOCIATED LITERATURE SOURCES ..... 35

# INTRODUCTION

Understanding and predicting the response of fish communities to habitat perturbations is a major challenge confronting freshwater ecologists. A suite of habitat conditions present at a variety of spatial and temporal scales can each play a potential role in influencing the distribution and abundance of fish species. Examining the importance of these individual features has traditionally employed non-mechanistic, correlative analyses with little reference to the biology of the species. Description and even prediction of ecological patterns may be achieved with these approaches but the results are likely to be of uncertain generality. It has been suggested that predictions of species distribution and response to changes in habitat should instead be based on a biologically integrated, mechanistic framework which examines the functional relationship between species and their environments (Poff 1997). Predictions of response to perturbation could then be much more robust and generalizable to a range of environmental conditions.

Orians (1980) argued that a framework of predictive community ecology could be constructed based on the response of species to abiotic environmental factors. For example, freshwater habitats with similar environmental selective forces should have fish species with similar adaptations (traits), based on elements of their physiology, behavior, or life history. These species traits presumably represent functional relationships with critical environmental variables, such as stream flooding or drying, local shear stress, temperature extremes, etc. that might be capable of inducing mortality (Poff 1997). A species persistence in a habitat is therefore based on its relative tolerance to a range of potentially changing environmental constraints. Predicting a response to habitat change consequently requires that we identify and quantify the key environmental features that represent selective forces capable of inducing mortality and assess the relative adaptive response of different species to these selective forces.

For the concept of species traits to be predictive, traits must be defined in terms of *functional significance* relative to qualitative or quantitative habitat conditions (Poff 1997). This functional significance can often be assessed *a priori*, based on biological understanding and theory. Examples might include an assessment that fast generation time is a key trait that promotes persistence in temporally unstable stream environments (Gray 1981), or that preferred spawning substrate may constrain reproductive success for certain fish species (Balon 1975). Determining how to group fish for other traits is often more difficult. Correlative approaches can be justified when species distribution and abundance are examined in response to strongly selective physiological gradients, for which an unambiguous mechanistic interpretation is available (Poff 1997). For example, Eaton et al. (1995) used a broad scale comparative study to establish relative differences in thermal tolerances among fish species. Experimental investigations of the functional significance of species traits can also help determine the importance of these traits in different environmental regimes. Smale and Rabeni (1995a, 1995b) experimentaliy determined hypoxia and hyperthermia tolerances in the laboratory for headwater stream fishes and used this information to successfully predict community composition among streams varying in dissolved oxygen and thermal characteristics. Examples of relevant species traits that have been characterized for fish include life history, trophic group, habitat preference, reproductive mode, body morphology, pollution tolerance, turbidity tolerance, thermal tolerance, hypoxia tolerance, and flood resistance (literature summarized in Poff 1997). Understanding the functional significance of species traits and examining multiple traits simultaneously can allow us to predict how fish species composition might change as environmental constraints vary spatially and temporally (Poff and Allan 1995; Rader 1997). Several authors have used species traits and environmental tolerances to determine the structure of stream fish assemblages in the western United States (e.g., Poff and Allan 1995; Hughes et al. 1998; Waite and Carpenter 2000), while

Parent and Schrimi (1995) synthesized information on species traits and associated ecological data to predict which fish species were most at risk of extinction in the Great Lakes region of eastern Canada.

Linkages between land use patterns and the functional significance of species traits can be used to establish the effect of environmental changes on community composition (Poff 1997). For example, if the effect of human activities (e.g., agriculture) was to greatly enrich streams, then pollution tolerance among the resident species should determine those best able to survive. Similarly, the effects of increased siltation would vary depending on relative tolerance to siltation and its associated impacts among the species. And if increased stream channelization from urbanization caused increased flashiness, then species with greater flood resistance would be more likely to survive. In this paper we employ this approach to predict which freshwater fish species should be most at risk from the impacts of logging in British Columbia. First, based on a review of the published literature, we determine the principal impacts to freshwater habitats caused by logging activities. We then identify generalized species traits most likely affected by each of these logging related impacts. Finally, using information from the published literature and expert opinion we create a matrix of sensitivity scores for species traits displayed by 91 freshwater fish species and subspecies known to exist in British Columbia. Marine species and those restricted to estuarine or tidally influenced waters were excluded from our assessments. We did include 15 freshwater species introduced into BC waters that have consequently established viable breeding populations in the province, including the Atlantic salmon (*Salmo salar*) for which successful breeding has only recently been documented (Volpe et al. 2000). The final composite value produced from the summed species trait scores represents a metric of relative logging risk among the freshwater fish taxa evaluated. This final score is used to rank BC's freshwater fish taxa for relative sensitivity to the combined effects of timber harvest.

# RATIONALE AND METHODS

## Forestry impacts on stream habitat

It is widely recognized that forestry practices have degraded water quality and fish habitat in western Canada and the northwestern United States (Slaney et al. 1977; Hartman and Scrivener 1990; Murphy 1995). Greater than 40% of BC's Provincial Forest has been harvested (Slaney and Zaldokas 1997) and most of the productive lowland streams have been logged to their stream banks. It is now evident that many past practices have resulted in impacts to freshwater habitats that will require extended periods for recovery and may represent significant threats to the sustainability of the provincial fisheries resource.

The primary effects of forest harvesting on freshwater habitats include:

1) increased sedimentation caused by landslides, debris flows, and bank erosion, as well as increased surface erosion and sediment transport from expanding road systems
2) increased stream temperatures
3) modified streamflow patterns
4) drastic simplification of in-stream habitat through loss of large woody debris (LWD) and associated pool-riffle structure
5) modification of riparian zones

6) changes in productivity (increasing or decreasing dependent on time scale)
7) restricted passage for adult or juvenile fish

These effects are examined in more detail below:

### *Sedimentation*

Sediments in streams and lakes originate either from surface erosion of exposed mineral soil or from mass movements such as landslides, debris torrents, slumps, and earthflows (Chamberlin 1991). In mountainous terrain, sediments of all sizes are delivered primarily by mass movements (Swanston 1991). These can occur as slow moving slumps and earthflows, or as episodic landslides and debris torrents which happen during heavy rainfall when saturated soils trigger slope failures (Murphy 1995). Incidences of landslides and debris flows increase with increased logging within watersheds, especially on steeper slopes in the coastal regions (Hogan 1986; Slaney and Zaldokas 1997). The increase in mass soil movements due to clear-cut logging varies widely, depending on the stability of the underlying bedrock and surficial glacial overburden, but can range from 2-4 times in Oregon and Washington (Ice 1985) to 31 times in the Queen Charlotte Islands of British Columbia (Rood 1984). Construction of roads will compound this situation (Jones et al. 2000), as landsliding associated with roads can be as much as 300 times more frequent than in undisturbed forest (Furniss et al. 1991). Mass wasting generally carries soils from hillsides to downstream depositional zones where the resultant increase in sediment may cause widening of mainstem channels, infilling of coarser substrates and blocking of side channels (Slaney and Zaldokas 1997). Adverse affects on fishes and other aquatic biota caused by these processes may persist far downstream for extended periods of time (Hicks et al. 1991a).

Typically, undisturbed forest soils will resist surface erosion because their coarse texture and thick layers of organic humus prevent overland flow (Murphy 1995). Extensive surface erosion can occur, however, in areas where soils have been exposed by major disturbances such as fire, landslides or clear-cutting (Swanston 1991). Transport of fine sediments from upslope banks, compounded by additional surface erosion from logging roads, can impair water quality, damage fish spawning gravels and rearing areas and decrease aquatic insect abundance (Slaney et al. 1977). Compaction of soil by logging equipment increases erosion by reducing surface infiltration and causing overland flow (Murphy 1995). Road surfaces, landings, skid trails, ditches and disturbed clear-cut areas can all contribute large quantities of fine sediments to streams (Reid and Dunne 1984; Chamberlin et al. 1991). Road construction and maintenance, log hauling, tree felling, yarding, slash disposal and silvicultural pre-treatments are all potential sources of fine sediment pollution (Chamberlin et al. 1991). Regular road use can cause chronic sediment inputs to streams nearly as great as those from original road construction. The road surface can break down with repeated truck traffic, resulting in a continual source of fine sediment (Murphy 1995). Increased levels of suspended sediment from disturbed soils and roads can persist for 6 - 10 years in coastal watersheds (Beschta 1978), but will last longer if the roads are still being used and maintained (Cederholm and Reid 1987).

Extensive accumulations of fine sediment in streams can affect fish directly as the long term survival for many fish species is dependent on the availability of silt-free gravel for spawning and rearing habitats (Beschta and Jackson 1979; Groot and Margolis 1991; Larkin and Slaney 1996; Larkin et al. 1998). Laboratory and mesocosm studies have demonstrated that high concentrations of suspended sediment can cause physical stress and mortality in sensitive fish species through direct physiological effects such as trauma to gill tissue or changes in blood

chemistry (McLeay et al 1987; reviewed in Newcombe and MacDonald 1991; Servizi and Martens 1991; reviewed in Anderson et al. 1996). Additionally, high levels of suspended sediment can adversely affect algal growth, biomass and species composition by reducing effective light penetration (Larkin et al. 1998) and may also clog benthos feeding structures (Larkin et al. 1998). Together these factors can ultimately result in an impoverished insect fauna (Newcombe 1985, Newcombe and McDonald 1991) potentially affecting the food production necessary to sustain fish populations. The slower growth rates documented for fish in turbid water (e.g., Sigler et al. 1984) may be a result of reduced abundance of macroinvertebrate prey or a reduction in the capability of the fish to find them, or both (Simmons 1984).

### *Temperature*

Streams are heated primarily by direct solar irradiation of the water surface and only minimally by convection or conduction (Beschta et al. 1987). Consequently, water temperatures are dictated principally by levels of shade rather than air temperatures. Removal of the tree canopy, therefore, can have dramatic effects on summer water temperatures raising them, in some cases, by as much as 10°C (Beschta et al. 1987). Although groundwater or hyporheic influences may modulate the effects (Shepard et al. 1986; Meisner et al. 1988), temperature increases are considered directly proportional to the area of stream opened to sunlight and inversely proportional to stream discharge (Beschta et al. 1987). Temperature increases are therefore potentially greatest in small shallow streams and diminish as streams get larger with a reduced influence of riparian canopy.

Major changes in water temperature can have serious consequences for cold-blooded organisms like fish which are adapted to a natural thermal regime. Temperature fluctuations can alter egg and juvenile fish development, inhibit growth, reduce survival and affect timing of life-history events (Beschta et al. 1987). High temperatures may also inhibit upstream migrations and increase disease (Hallock et al. 1970; Monan et al. 1975; Bjornn and Reiser 1991). Water temperatures will decline when regrowth of riparian vegetation again shades the stream. This can occur within 4 years in first-order streams but may take over 20 years in fourth-order streams (Beschta et al. 1987). Long-term warming of streams or lakes may result in range expansions for warmwater species, increasing predation and competitive interactions with coldwater fish (Reeves et al. 1987).

Potential temperature impacts on stream fish are likely to vary at a variety of scales both locally (dependent on stream gradient, depth, morphology, valley aspect, and bed materials) and regionally (dependent on latitude and climate) (Murphy 1995). Even in northern regions, however, high temperatures may be a problem for fish in "temperature sensitive" streams that may be wide, shallow, low gradient and have lake or muskeg sources (Gibbons et al. 1987). In southern regions with comparatively high ambient temperatures, increased stream temperatures may have significant impacts on some fish populations (Murphy 1995)

### *Streamflow Patterns*

Any precipitation not intercepted by plants will ultimately move downslope to enter stream channels as overland flow or infiltrate the soil to contribute to groundwater base flow (Swanston 1991). The larger the precipitation event, the greater the potential stream flow. As a watershed responds to precipitation, stream flow increases to maximum levels known as peak flows. High

or sustained rates of precipitation or snowmelt contribute to greater runoff and higher peak flows. Storm and watershed size, drainage network configuration, vegetation cover and topography influence the quantity of water and the rate at which it reaches stream channels and passes through the stream system (Swanston 1991). Interception and evotranspiration by plants can capture a large proportion of precipitation and plant roots anchor soil water from the pull of gravity. As a result, forest cover will retain some portion of any rainstorm and reduces the flood discharge and peak flows entering stream systems. Most undisturbed forest soils can accept water much faster than normal rates of rainfall or snowmelt, and water that reaches such soils generally infiltrates and follows subsurface routes to stream channels (Chamberlin et al. 1991). Substantial surface runoff normally occurs only when rainfall is particularly heavy.

Although forestry practices do not generally affect the total amount of precipitation entering a watershed, extensive harvesting can substantially alter the spatial distribution of water and snowpack on the ground, the amount of water intercepted or evaporated by foliage, the rate of snowmelt, the amount of soil water storage, and the physical structure of the soil that governs the rate of water movement to stream channels (Swanston 1991). Cutting trees can alter the overall water balance and accelerate movement of water from hillsides to stream channels. This can affect both the summer base flow and peak flows during rainstorms or snowmelt (Murphy 1995). Clear-cutting reduces a substantial area of leaves and stems that would otherwise intercept precipitation. Fewer tree roots additionally reduce the amount of water that can be extracted from the soil (Swanston 1991).

These factors cause soil water content and runoff to be higher in logged than in unlogged areas. Harvested areas contain wetter soils than unlogged areas and hence greater potential for late-summer runoff (Chamberlin et al. 1991). Summer flows have been shown to double or triple immediately after clear-cut logging in small watersheds (Rothacher 1971; Harr et al. 1979) and increases in annual runoffs of 30-40% post-harvest are not uncommon (Hibbert 1967). This effect generally lasts for about 3-5 years until new root systems occupy the soil (Chamberlin 1991), although it may take from 10-30 years for base flows to return to normal pre-harvest levels (Murphy 1995). In some situations base flows may decrease below pre-harvest levels because rapidly growing second-growth hardwoods transpire more water than mature conifer trees (Hicks et al. 1991b). In such cases decreased base flows can restrict the habitat available to fish, especially at critical low-flow periods in summer (Murphy 1995). Lowered base flow, combined with high temperatures and fine sediment inputs, can also deplete oxygen levels in a stream (Murphy 1995). Although dissolved oxygen (DO) in most natural streams is normally near saturation (about 10 mg/L at 10°C), local DO depletion as low as 2 mg/L has been documented for logged streams (Beshta et al. 1987). Oxygen concentrations at such reduced levels can induce hypoxia in fish, resulting in effects ranging from changes in behaviour and physiology to death of susceptible individuals (Davis 1975; Smale and Rabeni 1995a).

Road building, ditching, yarding, scarification and other forestry related activities can alter the pathways water takes to stream channels and typically shortens the routing time of water to the stream channel (Murphy 1995), the effects of which are often more evident in smaller streams than in larger rivers (Jones and Grant 1996). Surface exposure and compaction by logging equipment can also reduce the infiltration capacity of the soil, resulting in increased surface runoff and higher peak flows (Chamberlin et al. 1991; Jones et al. 2000). Hydrological effects associated with logging roads are likely to persist for as long as the road remains a physical feature altering flow routing, even persisting long after abandonment and revegetation of the road surface (Trombulak and Frissell 2000). Greater snow accumulation in clear cuts can also generate more severe rain-on-snow events and higher peak flows in winter and spring (Erman et al. 1988; Murphy 1995). Increases in peak flows can damage fish habitat as the resulting bedload

movement can deeply scour stream channels, killing incubating eggs (Seegrist and Gard 1972; Hartman and Scrivener 1990) or benthic-living fishes (Erman et al. 1988) and displacing other fish from cover (Tschaplinski and Hartman 1983). The ability of fish to withstand floods varies in relation to fish size and taxonomy but is primarily dependent on the timing and duration of spawning, as eggs and early life history stages are the most susceptible to the effects of erosive floods (Harvey 1987).

### *Riparian Zone*

The riparian zone is represented by the land adjacent to the normal high water line of a stream or lake, where soils and vegetation are influenced by the presence of the ponded or channelized water (Johnston and Slaney 1996). Small streams, in particular, are intimately connected physically, chemically and biologically to their riparian zone (Murphy and Meehan 1991). Roots of streamside vegetation stabilize stream banks, retard erosion, affect nutrients in groundwater, and create overhanging cover for fish (Murphy 1995). The riparian zone provides much of the future supplies of LWD to a stream by bank erosion and from windfall of mature trees (Murphy 1995). A canopy of riparian vegetation will contribute leaf detritus, terrestrial insects and other organic matter that is the food base for the stream ecosystem, and its shading moderates excessive increases in stream temperatures (Slaney and Zaldokas 1997).

The riparian zone also generally protects a mix of stream side-channels, off-channel ponds and access to small perennial tributaries which may serve as important overwintering habitat for fish (Murphy 1995; Slaney and Zaldokas 1997). Small intermittent streams are important in routing water, sediment and nutrients to downstream fish habitat (Murphy 1995). As intermittent streams can account for more than one-half of the total channel length in many small watersheds they can strongly influence the input of materials to the rest of the channel system (Murphy 1995). The influence of the riparian zone generally diminishes as streams get larger (Murphy and Meehan, 1991). Small streams receive more of their organic matter from local riparian vegetation and even small trees can provide effective shading. Large streams tend to receive more nutrients from secondary sources and even large trees may not effectively shade them (Murphy and Meehan 1991). The role of woody debris also changes with stream size. In small streams LWD debris may influence most of the stream channel, in larger streams debris is clumped into jams or pushed onto the banks (Murphy and Meehan 1991). Even in large rivers, however, LWD can create essential fish habitat by scouring deep holes in shallow river braids or creating side channels, off-channel sloughs, and marshes which increase habitat complexity and total rearing area (Murphy and Meehan 1991; Abbe and Montgomery 1996).

A past history of streamside logging to the banks has eliminated the original riparian zones of many BC streams, with serious impacts on bank stability and overwintering fish habitat (Slaney and Zaldokas 1997). Although removal of the riparian canopy may stimulate greater primary production due to increased light penetration (Murphy and Meehan 1991) the cumulative effects of increased water temperature and sediment from numerous disturbances in a watershed can nullify any beneficial effects of increased food production (Murphy and Meehan 1991). In some cases removing the canopy can cause stream temperatures to increase to lethal levels for fish, particularly salmonids (Hall and Lantz 1969). Logging to the stream banks has left most low-land streams in BC with primarily deciduous canopies, ranging up to 50 years old, or with immature conifer forests at higher elevations (Slaney and Zaldokas 1997). Deciduous trees provide little permanent LWD as windfalls, decay relatively quickly in water and also encourage beaver damming activity on small streams (Slaney and Zaldokas 1997). Although fish habitat

may initially be maintained by deciduous canopies, as the old-growth wood in the channel decays or shifts, the stream habitat becomes simple and monotypic, resulting in increased overwintering mortality for many fish species (Bisson et al. 1987; Slaney and Zaldokas 1997).

### *Large Woody Debris (LWD)*

Large woody debris (LWD) - defined as wood pieces at various stages of decay having a diameter 10 cm or larger over a minimum 2 m length and intruding into the bankfull channel (Johnston and Slaney 1996) - is an integral part of streams in forested watersheds. Large woody debris has an important role in stabilizing stream channels, providing structure to the stream, regulating storage of sediment and particulate organic matter and creating and maintaining fish habitat (Murphy 1995). Large woody debris enters streams via windthrow, streambank undercutting, or soil mass movement (Swanston 1991). Once in a stream, LWD may be deposited almost immediately within the channel or else transported downstream to various locations either singly or as jams of mixed logs and smaller organic debris. Large woody debris can change the longitudinal profile of a stream and reduce the local gradient of the channel (Swanston 1991). Stream channels with large amounts of LWD show more complex habitat structure with greater numbers of large, deep pools and an increased number of short, steep riffle sections relative to streams with limited LWD (Swanston 1991, Rosenfeld et al. 2000). Large jams of wood may sometimes block the passage of migrating fish and effectively close off areas to spawning for periods of time (Meehan 1974). Outweighing these occurrences, however, is the larger function that LWD serves in trapping and storing gravel necessary for the successful spawning of many fish species (Swanston 1991). The embedded wood also frequently creates a "stepped" channel profile, along which much of the gradient drop is over short wood-created falls that alternate with low-gradient pools. This gives much of the stream a lower gradient and creates more acceptable spawning and rearing habitat than might otherwise be available (Swanston 1991). Large woody debris also provides direct cover for resident and migrating fish and helps retain organic detritus entering the stream channel (Swanston 1991). This organic detritus provides the base for the food chain moving from microbes to insects to fish, and debris accumulations provide the necessary structures to locally retain these stream ecosystem components. Decreases in large pools and cover because of the loss of LWD may expose fish to predation and deprive them of critical habitat (Murphy 1995).

Past practices of logging forests to stream banks have greatly reduced the input of LWD from mature conifer windfalls, creating a serious long-term decline in the complexity and diversity of many stream systems (Slaney and Zaldokas 1997). Large pool frequencies, for example, decreased by nearly two-thirds after 50 years of logging history in Washington and Oregon streams (Murphy 1995). Logging activities can reduce LWD in several ways. Existing LWD can be destabilized during tree felling and yarding, and later exported downstream or onto the flood plain. (Bisson et al. 1987). Salvage of downed merchantable trees from the stream or cleaning of stream channels after yarding will remove LWD and destabilize channels (Bilby and Ward 1989). Removal of trees from the riparian area during logging also causes a long-term reduction in the recruitment of LWD. Young forest stands may not provide wood debris large enough to stabilize streams for at least 60-80 years after logging (Murphy and Koski 1989) or perhaps even longer if riparian areas are converted to non-conifer species (Murphy 1995). Timber harvest rotations of less than 100 years will permanently eliminate large LWD, unless streams are protected by buffer strips (Murphy 1995). Because almost half of BC's provincial forest has been logged, including the riparian zone of many low elevation streams, a serious LWD deficit continues to worsen (Slaney and Zaldokas 1997). It is expected that even in the absence of future logging activities

many of these streams will not recover naturally for one or two centuries, and existing fish habitat will continue to deteriorate for at least another 50 years (Slaney and Zaldokas 1997).

### *Stream Productivity*

Freshwater systems vary naturally in terms of their productivity, due to differences in nutrient and energy availability, and the rates of primary and secondary production will determine the amount of food ultimately available to fish (Bjornn and Reiser 1991). The primary production of many forested streams is often light limited. Removal of the riparian canopy by logging of streamside trees will increase light penetration, generally stimulating increased algal production within the stream (Murphy 1995). This increased algal production after canopy removal sometimes increases the abundance of invertebrates, mainly by enhancing the quality of detritus. When algae - a higher quality food than leaf or needle litter - detaches from the streambed, it enhances the quality of seston and deposited detritus (Hawkins and Sedell 1981; Hawkins et al. 1982). This enhanced detrital quality increases the abundance of invertebrate collectors and, in turn, increases the abundance of predatory invertebrates and fish (Murphy and Meehan 1991). Numerous studies have shown that unshaded streams will generally have greater invertebrate abundance than shaded streams (Chapman and Knudson 1980; Murphy and Hall 1981; Murphy et al. 1986). If key nutrients (i.e., phosphorus and/or nitrogen) remain scarce after disturbance, however, primary production may not increase even when the canopy is opened (Shortreed and Stockner 1983).

As riparian vegetation recovers after logging, the canopy closes again, light penetration is reduced, and aquatic production declines (Murphy and Meehan 1991, Murphy 1995). The tree canopy is generally denser in even-age, second growth forest and lacks the gaps that are common in old growth forest (Bjornn et al. 1992). Thus, increased primary and secondary productivity in the first 20 years after timber harvest may be followed by a much longer period of depressed production where fish biomass decreases to below pre-logging levels (Murphy and Meehan 1991).

A combination of past logging impacts and overfishing has also reduced the overall returns of salmonid spawners to BC river systems, resulting in potentially seriously impacts to the annual nutrient budgets of many streams and lakes (Slaney and Zaldokas 1997). Coastal and many interior streams in British Columbia are typically oligotrophic, with food bases that are phosphorus-nitrogen limited (Slaney and Zaldokas 1997). Recent studies have shown that returning salmon spawners contribute significant amounts of marine-derived nutrients (carbon, nitrogen, phosphorus) to natal waters and riparian zones (Bilby et al. 1996; Larkin and Slaney 1997). As such, salmonid carcasses are critical in providing many of the essential nutrients necessary to maintain aquatic food chains. Logging impacts that limit migratory fish returns (e.g., impassable road culverts, debris jams, reduced flow) could progressively remove a major nutrient-input into these freshwater systems (Slaney and Zaldokas 1997; Stockner et al. 2000).

### *Fish Passage*

British Columbia's fish fauna includes numerous anadromous species that undergo extensive migrations through freshwater systems during their lifecycles. Resident fish species may also move considerable distances up and downstream to seek food, shelter, better water quality, and spawning or overwintering areas (Furniss et al. 1991; Fausch and Young 1995). Movement of

fish can be blocked by waterfalls, debris jams, and other natural obstacles that can impede fish passage. Many of these barriers may be only temporary or dependent on seasonal flow conditions and obstacle configuration (Murphy 1995). Improperly engineered logging roads that cross streams, however, can create serious long-term problems for fish migration, usually because of outfall barriers, excessive water velocity, insufficient water depth in culverts, disorienting turbulent flow patterns, lack of resting pools below culverts, or a combination of these factors (Furniss et al. 1991). Some road crossings may act as semipermeable or seasonal barriers to fish movement, others may preclude all movement by fish, similar to the effects of dams (Warren and Pardew 1998). Elevated or steeply sloped road culverts are often impassable to fish, preventing upstream movements of adult spawners and eliminating access of juvenile fish to seasonal refuges from anchor ice and floods (Slaney and Zaldokas 1997; Trombulak and Frissell 2000). Such barriers to dispersal can delay or preclude recovery of fish assemblages following disturbance (Detenbeck et al. 1992) and increase the risk of metapopulation extinction by habitat fragmentation (Winston et al. 1991; Dunham and Rieman 1999). A single poorly installed culvert can eliminate the fish population of an entire stream system (Murphy 1995).

## Choice of functional species traits

Based on our review of logging impacts and a further assessment of the literature relating to fish physiology and ecology, we identified twelve species traits affecting fish response to timber harvest. The key functional traits we judged most critical in determining response were hyperthermia tolerance, hypoxia tolerance, siltation tolerance, rearing and spawning habitat use, reproductive strategy, migration strategy, spawning season, spawning mode, water column/substrate use, feeding mode, and use of large woody debris cover. Suggested linkages between these species traits and specific logging impacts are illustrated in Figure 1. Traits are discussed in greater detail below, as are expectations of relative sensitivities among species.

***Hyperthermia tolerance*** — Temperature tolerances can vary considerably among and within freshwater fish species (Hokanson 1977), and these tolerances will dictate natural distributions and response to elevated temperatures at different life cycle stages (Smale and Rabeni 1995a). Many fish species in BC are temperature sensitive, with narrow temperature requirements during spawning, incubation or rearing (Scott and Crossman 1973). Excessive warming of streams after canopy cover removal can make habitats unsuitable for cool or coldwater species (Bjornn and Reiser 1991). Fish will attempt to seek cold water refuges by moving up or downstream when water temperatures exceed thermal tolerances; rapid heating may, however, result in direct mortality (Bjornn and Reiser 1991). Coldwater species with narrow thermal tolerances are the fish most likely to be affected by changes in water temperature caused by the removal of protective canopies during timber harvest. This is particularly true for fish species resident in small streams where heating can be rapid and thermal refuges (e.g. deep pools) would be rare. Warmwater species or species with broad temperature tolerances would be less adversely affected, as would species in large rivers or lakes where increased temperature affects due to removal of the riparian zone would be minimal.

***Hypoxia tolerance*** — Dissolved oxygen (DO) content of surface waters can vary seasonally and spatially dependent on levels of photosynthesis, respiration, temperature, ice cover, turbulent mixing and input of organic materials requiring oxygen for aerobic decomposition (Barton and Taylor 1994). Similarly, oxygen requirements for efficient fish metabolism can vary dependent on season, age, health and life cycle stage. Low DO concentrations can cause hypoxia in fish,

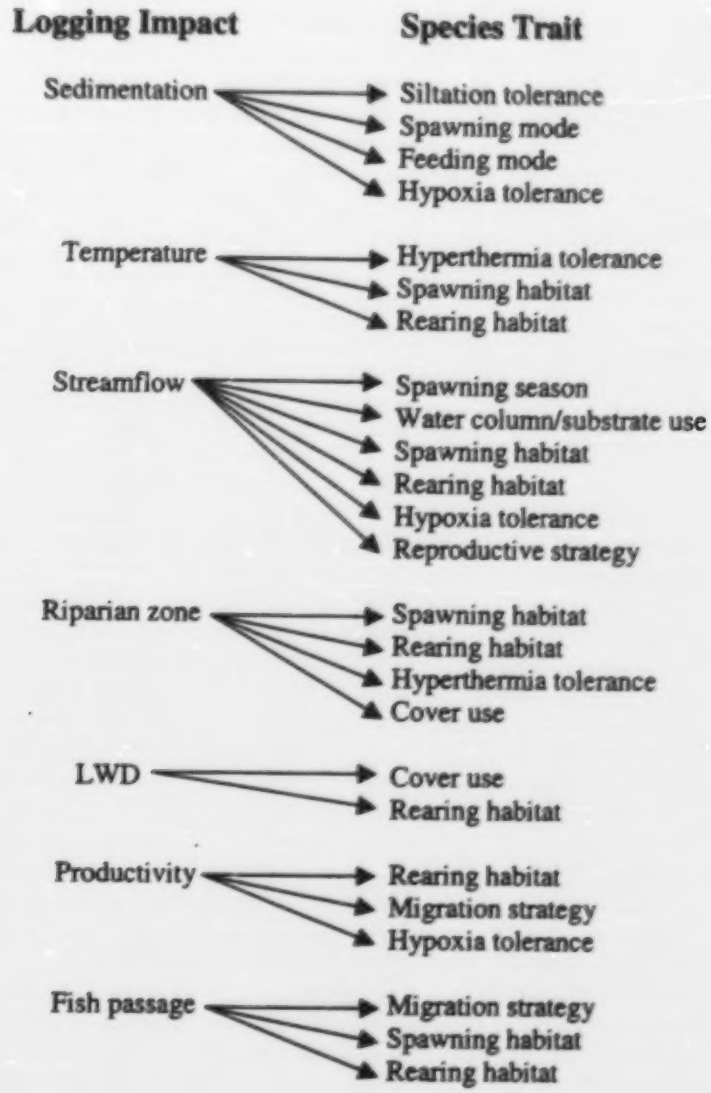


**Figure 1**. Suggested linkages between the impacts of logging on freshwater habitat and associated species traits.

a condition in which oxygen delivery to the cells is insufficient for normal metabolic needs (Doudoroff and Shumway 1970). Severe hypoxia can induce death from asphyxiation, whereas sublethal effects can include behavioural changes, alterations in blood chemistry, increased heart and ventilation rates, decreased disease resistance and reduced growth rates (Barton and Taylor 1994). Tolerances to hypoxia can vary considerably among fish species (Smale and Rabeni 1995b) and these differences may dictate their use of habitats and response to logging-induced changes in physiochemical conditions. For example, a decrease in DO from 8.7 mg/L to 4.1

mg/L downstream of an impounded river resulted in loss of two sensitive trout species and a major restructuring of the remaining fish community (Camargo 1991). Fish species typically inhabiting fast, clean rivers or cold, clear lakes are expected to have high DO requirements and limited tolerances for hypoxia (Barton and Taylor 1994). Salmonids in particular are known to experience sublethal effects at DO concentrations as high as 6 mg/L, with onset of lethal effects occurring at concentrations near 2 mg/L (Davis 1975). Highly tolerant species (in which acute lethal limits may be well below 1 mg/L) commonly inhabit shallow ponds and backwaters or frequent aquatic vegetation along riverbanks and lakeshores, where natural diurnal fluctuations in DO are apt to be severe (Barton and Taylor 1994).

***Habitat use*** — Fish species most likely at risk from logging are stream-dependent for at least one stage of their life cycle, particularly in headwater/small stream situations where they may also be isolated and distinct (Haas 1998). This relates to the greater immediate impact that logging effects (e.g. increased erosion, modified runoff, altered riparian vegetation, etc.) have on smaller versus larger streams (Murphy 1995). Fish resident in lakes are not immune to the detrimental effects of forest harvest as riparian deforestation can result in deepened thermoclines, consequently restricting the availability of well-oxygenated hypolimnetic water for deepwater species like lake trout (France 1997). Lake fish may also use LWD from riparian sources as important cover structure in nearshore areas (Miller et al. 1997). However, it is generally assumed that logging of adjacent forests has less influence on lakes than on streams. The least sensitive fish would reside principally in large rivers where riparian forests have minimal influence.

***Siltation tolerance*** — Increased input of fine sediment into streams and lakes is a well-documented result of logging and associated road building activities (Murphy 1995). Response of fish to suspended sediment is a function of sediment concentration and duration of exposure (Newcombe and MacDonald 1991). Dependent on these factors and relative species tolerances direct effects of fine sediment on fish can range from changes in behaviour (e.g., abandonment of cover, alarm reaction), to sublethal effects (e.g., reduction in feeding rate, moderate physiological stress, increased respiration rate, impaired homing) to lethal and paralethal effects (e.g., reduced growth rate, delayed hatching, increased predation, 20 - 100% mortality: summarized in Newcombe and Jensen 1996). Certain fish species and/or lifestages may be ultrasensitive to increased levels of fine sediment, with abrupt onset of detrimental effects above certain threshold concentrations (Newcombe and Jensen 1996). Fish species typically dependent on clear, gravelly streams are likely to be sensitive to increased concentrations of fine sediment, whereas species displaying a mix of habitat use are probably more capable of withstanding influxes of fine sediment, particularly if only of short duration. Fish species commonly inhabiting slow water, silted rivers and lakes are likely to be highly tolerant of fine sediment and therefore should be less affected by increased sediment loads. Many lake resident species spawn, however, in small tributary streams and their eggs and juveniles may not be as resilient to the effects of increased siltation.

***Reproductive strategy*** — The longer term consequences of habitat perturbation to freshwater communities may be buffered for fish species which can quickly reestablish population densities after disturbance, a characteristic which has been termed population "resilience" (Harrison 1979; Townsend and Hildrew 1994). These are populations which, despite often serious declines in the short-term, can rebound quickly to their former state. Species resilience can be anticipated from individual reproductive traits that determine the return time of populations via population growth and the rate of generational turnover, an argument based on the same general reasoning as that associated with the r/K concept of Pianka (1970). The key

reproductive traits in this regard include age of first reproduction, potential life span, body size, number of reproductive efforts per season and number of reproductive cycles per lifespan (summarized in Townsend and Hildrew 1994). Highly resilient species would be expected to have small body sizes, short life spans, early maturity, and possibly repeated spawning efforts within a single season. Conversely, species with limited resilience should generally have large body sizes, long life spans, delayed maturity, and a single spawning effort (or less) per season. Fish species with high individual fecundity could, however, offset other reproductive traits that might imply poor resiliency. Species of salmonids, sturgeons and lampreys (Scott and Crossman 1973), for example, can produce copious amounts of eggs and should be capable of repopulating disturbed areas quite rapidly if access and habitat quality is restored. This strategy is unlikely to be successful, however, for species that are faced with a repeated series of serious habitat impacts over an extended time frame (Rose 2000).

***Feeding mode*** — Fine sediment inputs from timber harvest and logging road construction are a primary source of increased turbidity to otherwise naturally clear-water habitats. It has been suggested that turbidity is the most predominant adverse water quality characteristic of human-impacted streams, affecting approximately 34% of streams in the United States, for example (Lloyd et al. 1987). Increased turbidity has been shown to hamper the ability of fish that feed by sight to successfully capture food, due to reduced levels of available light (Lloyd et al. 1987.) Reactive distances, feeding success, and feeding rates may all be reduced for fish in turbid waters (Gardner 1981; Schmidt and O'Brein 1982; Crecco and Savoy 1984; Gregory and Levings 1998). Increased siltation can provide a source of beneficial cover for larval fish (Gregory 1993) and predation by piscivores is often reduced under conditions of high turbidity (Gregory and Levings 1998; Johnson and Hines 1999). However, most fish species will avoid turbid areas of streams or if forced to remain in such areas often grow slower than fish in clearer streams in the near vicinity (Lloyd et al. 1987). Reduced light availability caused by high turbidities also can limit the production and abundance of phytoplankton in clear streams or lakes (Murphy 1995), thereby reducing the food supply for algivores. The fish most sensitive to increased turbidity, therefore are sight-dependent feeders (piscivores, planktivores or invertebrate drift feeders) or algivorous species feeding in clear streams or lakes. Species most robust to changes in turbidity could be either non-visual feeders dependent on chemical or tactile clues for locating prey or those with vision pre-adapted for locating food in turbid conditions.

***Spawning mode*** — Logging and associated activities can increase transport of fine sediments into spawning and rearing areas for fish (Murphy 1995). Species that require clean substrate for spawning have been shown to be negatively affected by siltation in streams (e.g., Berkman and Rabeni 1987). Fish best able to cope with this sediment influx must either be able to keep their eggs clean of silt or be pre-adapted for successful spawning in silted environments. Such species generally exhibit a high level of parental care that often involves guarding and fanning the eggs and may typically spawn in silted areas amongst vegetation. Species most likely at risk are broadcast spawners who release their eggs over areas of clean rocky or gravel substrate and then abandon the eggs (defined as simple, lithophilous spawners (Berkman and Rabeni 1987)). Other lithophilous species requiring clean spawning gravels (e.g. salmonids) display some level of prespawning behaviour (e.g. redd construction) that will clear an area free of silt (Bjornn and Reiser 1993). However, there is limited post-spawning behaviour in these cases to maintain the quality of the nesting site for the duration of egg development.

***Migration strategy*** — Fish in freshwater systems may undertake several different types of migration. These include diadromy, or movement between fresh and seawater habitats, (with anadromy referring to migration from the sea up rivers to spawn in freshwater and catadromy

referring to migrating down rivers to spawn in seawaters) and potodromy, movement solely within freshwater (Northcote 1997). Within potodromous populations there is also variation. Some species confine their cycles of seasonal trophic and reproductive migrations within a single stream or river drainage (fluvial migration). Other species move between main stem rivers and smaller tributary streams for spawning, feeding or survival (fluvial-adfluvial migration). Still others spend most of their life history within lakes but may ascend small inlet or outlet streams for spawning purposes (lacustrine-adfluvial populations; summarized in Northcote 1997). Barriers to movement can affect the long-term survival of fish populations displaying any of these migration traits, particularly if access to suitable spawning habitats is restricted. Logging debris and associated road construction activities can seriously impede movements of migrating fish, particularly in small streams (Murphy 1995). Fish most likely to be affected by logging activities are those undertaking longer journeys with a consequent greater potential to meet significant barriers. This is especially true for species moving through small stream systems. Fluvial populations displaying only limited movements in small streams or diadromous species moving only through larger river systems (where barriers related to logging activities are minimal) are likely to be less affected. Lacustrine populations spawning in small tributary streams may be at risk, but fish species whose movements are confined solely to lake waters should be relatively unimpeded by logging activities.

***Water column/substrate use*** —Removal of upland forest through timber harvest results in increased surface runoff and higher peak flows in streams (Murphy 1995). Rapid transport of stream bed materials (e.g., boulders, rocks) in flood events has been shown to directly kill many benthic-living fishes resident in streams amongst rocky substrates (Erman et al. 1988). Although flooding events may temporarily displace other fish species from preferred cover (Tschaplinski and Hartman 1983), the direct impact on non-benthic species is likely much less severe. Benthic species resident in lakes or living in softer sand and mud substrates are likely also at less direct risk from increased flooding events.

***Spawning season*** — Clearcut logging increases peak flow rates in streams for both spring and winter floods (Murphy 1995). BC's coastal streams generally have highest flow in fall and winter because of heavy rain and wet snow, whereas in the interior stream flows are highest in spring during snowmelt (Murphy 1995; Montgomery et al. 1999). Although peak flow in interior streams may be greater during spring, winter flooding can actually be more severe because accumulated snowpack (which is greater if the tree canopy has been removed) increases the effective height of the streambank and confines most or all of a rain-on-snow flood within the channel (Erman et al. 1988). This produces greatly increased shear forces on the streambed which moves bed-material along rapidly, resulting in the destruction of the buried eggs of fall-spawning fishes by mechanical grinding or crushing (McNeil 1964; Tripp and Poulin 1984; Erman et al. 1988; Hartman and Scrivener 1990). Conversely, high discharge during snow-free periods (spring/summer) generally spills into the flood plain, minimizing additional increases in stream depth and shear force (Erman et al. 1988). It seems likely that only extremely rare floods would be large enough to affect fish populations when flow is not constrained by an existing snowpack along the channel banks (Erman et al. 1988). As such, the eggs of spring and summer-spawning fish should be relatively safer from the effects of logging on peak flows. Additionally, the relatively longer duration of the intergravel life stage of fall spawners increases their exposure to logging related episodes of fine sediment deposition and other detrimental changes in water quality than might be the case for spring and summer-spawning fish.

***Cover use*** — Cover is an important component of fish habitat, essential in providing security from predation and allowing fish to occupy areas of streams and lakes that might

otherwise be unavailable. Some of the features that may provide cover for fish include water depth, turbulence, cobbles and boulders, overhanging or undercut banks, overhanging riparian vegetation, small or large woody debris, and aquatic vegetation (Bjornn and Reiser 1991). Fish abundance in streams is often positively correlated with the abundance and quality of cover (Bjornn and Reiser 1993). Large woody debris originating from riparian forest can be the primary source of cover for many species of fish (e.g., Lehtinen et al. 1997), particularly for salmonids in small streams (Dolloff 1986; Elliott 1986; Bjornn et al. 1993). Other species of fish show greater use of alternative cover sources (e.g., Cunjak 1995) and may not be directly affected if logging activities reduce the normal input of LWD.

## Derivation of species trait information

Information for individual fish taxa was derived, whenever possible, from existing overviews of functional traits published in the literature. For example, experimental and field information on direct lethal and sublethal response to high temperatures, low dissolved oxygen levels and increased sediment concentrations have been summarized for many temperate fish species in various government reports and in the primary literature (e.g., Black 1953; Doudoroff and Shumway 1970; Coutant 1977; Davis 1975; Hokanson 1977; Ford et al. 1992; Barton and Taylor 1994; Eaton et al. 1995; Smale and Rabeni 1995b; Anderson et al. 1996). Reproductive trait data has additionally been summarized and quantified for many BC fish species within FishBase 99 (http://www.fishbase.org), an interactive web-based source for evaluating information on global fisheries. Rearing and spawning habitats for each species, feeding and spawning modes, migration strategies, use of cover and substrate, and migration strategies were determined, wherever possible, from status reports on individual species (e.g., Moodie 1984; Peden and Hughes 1984; Beamish 1987; McPhail 1987; Hughes and Peden 1989; Lane 1991) or from reports of directed research on a particular species (e.g., Fuller 1955; Winn 1960; Dryfoos 1965; Hauser 1969; Dauble 1986; Pearson 1998; McPhail and Baxter 1996; Rosenfeld et al. 2000). In other cases this data was extracted from general references describing the habitats and life history characteristics of fish in BC waters (e.g., Scott and Crossman 1973; Wooton 1984; Groot and Margolis 1991; Smith 1991; Cannings and Ptolemy 1998) or from general research on a life history trait that could be applied to selected species (e.g., Erman et al. 1988; Lloyd et al. 1988; Poff and Allan 1995; Cunjak 1996; Lehtinen et al. 1997; Northcote 1997; Johnston 1999).

Fish species can show flexibility in their functional traits and often undergo ontogenetic shifts in many life history characteristics (Poff and Allan 1995). We based our categorizations principally on the ecology of adult life stages, and made general assessments of each species trait based on a synthesis of multiple literature sources whenever possible. Although some species can show considerable breadth in their habitat use and life history parameters, we tried to ascertain the principal strategies employed within a species. We attempted to find at least two published references (with generally a maximum of five) relating to specific traits for each species. In cases where information varied between sources we attempted either to establish the most common interpretation of the trait, or used provincial or regionally based information if available. Published descriptions of some functional traits was lacking for many species (particularly non-game species). In these cases we based our interpretations of specific traits on general life-history descriptions or used information from related species if available.

## Scoring of species traits using sensitivity categories

Each functional trait category was scored for a taxa based on broad sensitivity criteria defined in Table 1. Categories for which finer scale data was available were subdivided into three scoring classes (1, 2 or 3), other traits with more limited datasets were scored as only 1 or 2. A scoring of 1 indicated low sensitivity to a logging related impact, 2 indicated relatively higher sensitivity (moderate or high), and 3 indicated highest sensitivity for a species. Species trait categories and scores were developed by the primary author based on available literature. Results were submitted for evaluation to G. Haas (BC Fisheries, Fisheries Research Section) an expert fish biologist familiar with BC's regional ichthyofauna. This expert opinion gave support to the general species trait characterizations and provided additional information for species where published data was lacking or of limited quality.

A matrix of functional trait scores was then constructed for the 91 fish species and subspecies evaluated (Table 2). Scores were summed as unweighted numeric values, and totals for each species were input into a *k*-means clustering analysis using Systat (1996). *K*-means clustering splits a set of values into a selected number of groups by maximizing between-cluster variation relative to within-cluster variation (Wilkinson et al. 1996). It is similar to a one-way analysis of variance where the groups are unknown and the largest F value is sought by reassigning members to each group (Wilkinson et al. 1996). A desirable feature of the clustering analysis is that it allows the user to set the number of ecologically similar groups to be identified. We set the procedure to identify three distinct clusters of fish taxa based on trait scores, which would be characterized as being low, moderate or high sensitivity to logging impacts. All clustering methods have analytical limitations (Milligan 1980), and should best be thought of as descriptive methods for pattern analysis (Poff and Allan 1995). Given both the relatively qualitative nature of our input data and our general goal of discerning broad variation in species sensitivity we felt the method best suited for our analyses. It should be noted that our categorizations provide only a measure of relative sensitivity among the species, and it should not be inferred that species within the lowest sensitivity category are immune from all detrimental effects associated with logging.

# RESULTS

Our clustering into three ranked groups based on summed attribute scores placed 26 taxa into a highest sensitivity category (total scores ranging from 24 to 29, mean = 25.4), 43 taxa into a moderate sensitivity category (scores ranging from 19 to 23, mean = 21.0), and 22 taxa into a lowest sensitivity category (scores ranging from 13 to 18, mean = 16.1) (Table 2). Our analysis would indicate that approximately 76% of BC's freshwater fish taxa are at least moderately sensitive to the effects of timber of harvest and would suggest that about 29% could be considered highly sensitive.

Sensitivity was generally high amongst salmonids, with 74% (20/27) of the evaluated taxa scoring in the highest sensitivity category, while no salmonid taxa grouped into the lowest sensitivity category. The sole gadid species found in British Columbia waters (burbot) scored within the highest sensitivity category. Catostomids (33%, 2/6), cottids (25%, 2/8) and petromyzids (20%, 1/5) also had taxa displaying the highest sensitivity ratings. Among these families only the Cultus Lake sculpin scored for lowest sensitivity to logging impacts; minimum scoring for other taxa within these families was at least for moderate sensitivity. All osmeridae species (3/3), both species of sturgeon (acipenseridae) and the sole BC representatives of the clupeid and percopsid families (American shad and troutperch respectively) were grouped into a

**Table 1.** Functional species traits affecting the sensitivity of fish to the impacts of logging on freshwater habitats. A scoring of 1 indicates the lowest rating of sensitivity for a species and 2 (and then in some cases 3) indicates higher levels of sensitivity.

(A) Hyperthermia tolerance

1. Tolerant of high water temperatures (temperate eurytherm with upper lethal temp > 34°C) or found predominantly in warm water
2. Moderate tolerance for high water temperatures (temperature mesotherm with upper lethal temp 28-34°C) or found predominantly in cool water
3. Low tolerance for high water temperatures (temperature stenotherm with upper lethal temp < 26°C) or found predominantly in cold water

(B) Habitat use-Spawning

1. Spawns predominantly in lakes or medium to large sized rivers
2. Spawns predominantly in small to medium sized rivers
3. Spawns predominantly in small creeks and headwaters

(C) Habitat use-Rearing

1. Occurs predominantly in large rivers or lakes
2. Occurs predominantly in medium to large sized rivers
3. Occurs predominantly in small creeks and headwaters

(D) Hypoxia tolerance

1. Tolerant of low DO concentration (lethal limit < 1 mg/L0 or generally inhabits eutrophic systems
2. Intermediate sensitivity to low DO concentrations (lethal limit 1-2 mg/L) or generally inhabits mesotrophic systems
3. High sensitivity to low DO concentrations (lethal limit > 2mg/L) or generally inhabits oligotrophic systems

(E) Siltation tolerance

1. Physiologically tolerant of high levels of suspended sediment or commonly found in silted waters
2. Physiologically intolerant of high levels of suspended sediment or commonly found only in clear waters

(F) Reproductive strategy

1. Short life span, early age of sexual maturity (generally in 1$^{st}$ or 2$^{nd}$ year), fast generation times (1-3 years) - rapid return time of populations after disturbance
2. Moderate life span, intermediate age of sexual maturity ( generally in 3$^{rd}$ to 4$^{th}$ year), moderate generation times (3 - 6 years) - intermediate return time of populations after disturbance
3. Long life span, delayed age of sexual maturity (generally in 5$^{th}$ year or later), slow generation times (> 6 years) - very slow return time of populations after disturbance despite often high individual fecundity

(G) Feeding mode

1. Nonvisual predator/scavenger/omnivore, eyes adapted for reduced light levels, or else does not feed in freshwater
2. Visual predator/planktivore reliant on good light for efficient feeding, or else algivore

(H) Spawning mode

1. Spawns successfully on miscellaneous substrate (sand, silt, vegetation), often with parental care
2. Spawns on clean gravel/rock substrate (lithophilous) with pre-spawning site preparation or parental care of eggs
3. Spawns on clean gravel/rock substrate (lithophilous) without site preparation (simple broadcast spawner)

(I) Migration patterns

1. Limited movements through stream or river systems for spawning or overwintering purposes
2. Moderate movements through stream or river systems
3. Extensive movements through stream or river systems

(J) Water column /substrate use

1. Primarily living within the water column or is benthic living on soft substrate
2. Primarily benthic living amongst rocky substrate

(K) Spawning season

1. Spring or summer spawner (relatively less exposure to intense flow events)
2. Fall spawner (relatively greater exposure to intense flow events)

(L) Cover use

1. Limited use of large woody debris cover by juveniles and/or adults
2. Extensive use of large woody debris cover by juveniles and/or adults

**Table 2.** List of 91 freshwater fish species and subspecies found in British Columbia waters and their associated functional trait scores in relation to potential logging impacts (T = temperature tolerance, HS = spawning habitat, HR = rearing habitat, O2 = hypoxia tolerance, ST = siltation tolerance, RS = reproductive strategy, FM = feeding mode, SM = spawning mode, MS = migration strategy, WC = water column/substrate use, SS = spawning season, C = cover use, Tot = total score for the species, Cat = sensitivity category - low(L), medium(M) or high(H). See table 1 for scoring criteria.

| Family/taxa | T | HS | HR | O2 | ST | RS | FM | SM | MS | WC | SS | C | Tot | Cat |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Acipenseridae | | | | | | | | | | | | | | |
| *Acipenser medirostri* | 2 | 1 | 1 | 2 | 1 | 3 | 1 | 3 | 2 | 1 | 1 | 1 | 19 | M |
| *A. transmontanus* | 2 | 1 | 1 | 2 | 1 | 3 | 1 | 3 | 2 | 1 | 1 | 1 | 19 | M |
| Clupeidae | | | | | | | | | | | | | | |
| *Alosa sapidissima* * | 1 | 1 | 1 | 2 | 2 | 3 | 2 | 1 | 3 | 1 | 1 | 1 | 19 | M |
| Centrarchidae | | | | | | | | | | | | | | |
| *Lepomis gibbosus* * | 1 | 1 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 1 | 1 | 1 | 16 | L |
| *Micropterus dolomieu* * | 1 | 1 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 1 | 1 | 2 | 17 | L |
| *M. salmoides* * | 1 | 1 | 1 | 1 | 2 | 2 | 2 | 1 | 1 | 1 | 1 | 2 | 16 | L |
| *Pomoxis nigromaculatus** | 1 | 1 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 1 | 1 | 2 | 17 | L |
| Cottidae | | | | | | | | | | | | | | |
| *Cottus aleuticus* | 3 | 3 | 3 | 2 | 2 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 24 | H |
| *C. asper* | 2 | 2 | 2 | 2 | 2 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 21 | M |
| *C. bairdi* | 3 | 2 | 2 | 3 | 2 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 23 | M |
| *C. cognatus* | 3 | 2 | 2 | 2 | 2 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 22 | M |
| *C. confusus* | 3 | 3 | 3 | 2 | 2 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 24 | H |
| *C. ricei* | 3 | 2 | 2 | 2 | 1 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 21 | M |
| *C. rhotheus* | 3 | 2 | 2 | 3 | 2 | 1 | 2 | 2 | 2 | 2 | 1 | 1 | 23 | M |
| *C. sp. - Cultus Lake sculpin* | 3 | 1 | 1 | 2 | 2 | 1 | 2 | 2 | 1 | 1 | 1 | 1 | 18 | L |
| Cyprinidae | | | | | | | | | | | | | | |
| *Acrocheilus alutaceus* | 1 | 2 | 2 | 2 | 2 | 2 | 2 | 3 | 2 | 1 | 1 | 1 | 21 | M |
| *Carrassius aurateus* * | 1 | 1 | 1 | 1 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 1 | 13 | L |
| *Couesius plumbeus* | 1 | 2 | 2 | 2 | 1 | 2 | 2 | 3 | 2 | 1 | 1 | 2 | 21 | M |
| *Cyprinus carpio* * | 1 | 1 | 1 | 1 | 1 | 3 | 1 | 1 | 1 | 1 | 1 | 2 | 15 | L |
| *Hybognathus hankinsoni* | 2 | 3 | 3 | 2 | 1 | 1 | 2 | 1 | 2 | 1 | 1 | 1 | 20 | M |
| *Margariscus margarita* | 2 | 2 | 2 | 2 | 2 | 1 | 2 | 3 | 2 | 1 | 1 | 1 | 21 | M |
| *Mylocheilus caurinus* | 2 | 1 | 1 | 2 | 2 | 2 | 2 | 3 | 2 | 1 | 1 | 1 | 20 | M |
| *Notropis atherinoides* | 2 | 1 | 1 | 2 | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 16 | L |
| *N. hudsonius* | 2 | 1 | 1 | 2 | 2 | 1 | 2 | 3 | 1 | 1 | 1 | 1 | 18 | L |
| *Phoxinus eos* | 1 | 2 | 2 | 2 | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 17 | L |
| *Phoxinus neogaeus* | 1 | 2 | 2 | 2 | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 17 | L |
| *Pimephales promelas* * | 1 | 2 | 2 | 1 | 1 | 1 | 2 | 1 | 1 | 1 | 1 | 2 | 16 | L |
| *Platygobio gracilis* | 2 | 2 | 1 | 2 | 1 | 2 | 2 | 3 | 2 | 1 | 1 | 1 | 20 | M |
| *Ptychocheilus oregonensis* | 1 | 1 | 1 | 2 | 2 | 2 | 2 | 3 | 2 | 1 | 1 | 1 | 19 | M |
| *Rhinichthys cataractae* | 2 | 2 | 2 | 2 | 2 | 1 | 2 | 3 | 2 | 2 | 1 | 1 | 22 | M |
| *Rhinichthys osculus* | 1 | 2 | 1 | 2 | 2 | 1 | 2 | 3 | 2 | 2 | 1 | 1 | 20 | M |
| *Rhinichthys falcatus* | 1 | 1 | 1 | 2 | 1 | 1 | 2 | 3 | 2 | 2 | 1 | 1 | 18 | L |
| *Rhinichthys sp.- Umatilla dace* | 2 | 1 | 1 | 2 | 2 | 1 | 2 | 3 | 2 | 2 | 1 | 1 | 20 | M |
| *Rhinichthys sp.*-Nooksack dace | 2 | 3 | 3 | 2 | 2 | 1 | 2 | 3 | 1 | 2 | 1 | 1 | 23 | M |
| *Richardsonius balteatus* | 1 | 2 | 2 | 2 | 2 | 1 | 2 | 3 | 2 | 1 | 1 | 1 | 20 | M |
| *Tinca tinca* * | 1 | 1 | 1 | 1 | 1 | 3 | 1 | 1 | 1 | 1 | 1 | 1 | 14 | L |
| Catostomidae | | | | | | | | | | | | | | |
| *Catostomus catostomus* | 3 | 2 | 2 | 3 | 1 | 3 | 1 | 3 | 2 | 1 | 1 | 1 | 23 | M |
| *C. columbianus* | 2 | 2 | 2 | 2 | 1 | 2 | 1 | 3 | 2 | 1 | 1 | 1 | 20 | M |
| *C. commersoni* | 2 | 2 | 2 | 2 | 1 | 3 | 1 | 3 | 2 | 1 | 1 | 1 | 21 | M |
| *C. macrocheilus* | 2 | 2 | 1 | 2 | 1 | 3 | 1 | 3 | 2 | 1 | 1 | 1 | 20 | M |
| *C. platyrhynchus* | 3 | 3 | 3 | 3 | 1 | 2 | 1 | 3 | 2 | 2 | 1 | 1 | 25 | H |
| *C. sp.* (Salish sucker) | 2 | 3 | 3 | 3 | 2 | 3 | 1 | 3 | 2 | 1 | 1 | 1 | 25 | H |
| Esocidae | | | | | | | | | | | | | | |
| *Esox lucius* | 2 | 1 | 1 | 1 | 2 | 3 | 2 | 1 | 1 | 1 | 1 | 1 | 17 | L |

| Family/taxa | T | HS | HR | O2 | ST | RS | FM | SM | MS | WC | SS | C | Tot | Cat |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Gadidae | | | | | | | | | | | | | | |
| *Lota lota* | 3 | 1 | 1 | 3 | 2 | 3 | 2 | 3 | 2 | 2 | 2 | 2 | 25 | H |
| Gasterosteidae | | | | | | | | | | | | | | |
| *Culaea inconstans* | 2 | 2 | 2 | 2 | 2 | 1 | 2 | 1 | 2 | 1 | 1 | 2 | 20 | M |
| *Gasterosteus aculeatus* | 1 | 2 | 2 | 2 | 2 | 1 | 2 | 1 | 3 | 1 | 1 | 2 | 20 | M |
| *G. sp.*- unarmoured stickles | 2 | 1 | 1 | 2 | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 16 | L |
| *G. sp.* - lake stickles | 2 | 1 | 1 | 2 | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 16 | L |
| *G. sp.* - giant black stickles | 2 | 1 | 1 | 2 | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 16 | L |
| *Pungitius pungitius* | 2 | 2 | 2 | 2 | 1 | 1 | 2 | 1 | 2 | 1 | 1 | 2 | 19 | M |
| Hiodontidae | | | | | | | | | | | | | | |
| *Hiodon alosoides* | 2 | 1 | 1 | 2 | 1 | 3 | 1 | 1 | 3 | 1 | 1 | 1 | 18 | L |
| Icturalidae | | | | | | | | | | | | | | |
| *Ameiurus melas* * | 1 | 1 | 1 | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 1 | 14 | L |
| *A. nebulosus* * | 1 | 1 | 1 | 1 | 1 | 2 | 1 | 1 | 1 | 1 | 1 | 2 | 14 | L |
| Osmeridae | | | | | | | | | | | | | | |
| *Spirinchus thaleichthys* | 3 | 1 | 1 | 3 | 2 | 1 | 2 | 3 | 3 | 1 | 2 | 1 | 23 | M |
| *S. sp.* - pygmy longfin smelt | 3 | 1 | 1 | 3 | 2 | 1 | 2 | 3 | 2 | 1 | 2 | 1 | 22 | M |
| *Thaleichthys pacificus* | 3 | 1 | 1 | 3 | 2 | 1 | 1 | 3 | 3 | 1 | 1 | 1 | 21 | M |
| Percidae | | | | | | | | | | | | | | |
| *Perca flavescens* * | 2 | 1 | 1 | 1 | 2 | 2 | 2 | 1 | 1 | 1 | 1 | 1 | 16 | L |
| *Stizostedion vitreum* | 2 | 1 | 1 | 2 | 1 | 3 | 1 | 3 | 2 | 1 | 1 | 2 | 20 | M |
| Percopsidae | | | | | | | | | | | | | | |
| *Percopsis omiscomaycus* | 3 | 2 | 1 | 2 | 1 | 1 | 2 | 3 | 2 | 1 | 1 | 1 | 20 | M |
| Petromyzidae | | | | | | | | | | | | | | |
| *Lampetra ayresi* | 3 | 1 | 1 | 3 | 1 | 3 | 2 | 2 | 3 | 1 | 1 | 1 | 22 | M |
| *L. macrostoma* | 3 | 1 | 1 | 3 | 1 | 3 | 2 | 2 | 1 | 1 | 1 | 1 | 20 | M |
| *L. richardsoni* | 3 | 2 | 2 | 3 | 1 | 3 | 2 | 2 | 1 | 1 | 1 | 1 | 22 | M |
| *L. richardsoni marifuga* | 3 | 3 | 3 | 3 | 1 | 3 | 2 | 2 | 1 | 1 | 1 | 1 | 24 | H |
| *L. tridentata* | 2 | 2 | 2 | 3 | 1 | 3 | 2 | 2 | 3 | 1 | 1 | 1 | 23 | M |
| Salmonidae | | | | | | | | | | | | | | |
| *Coregonus artedii* | 3 | 1 | 1 | 3 | 2 | 2 | 2 | 3 | 1 | 1 | 2 | 1 | 22 | M |
| *C. autumnalis* | 3 | 1 | 1 | 3 | 2 | 3 | 2 | 3 | 2 | 1 | 2 | 1 | 24 | H |
| *C. clupeaformis* | 3 | 1 | 1 | 3 | 2 | 2 | 2 | 3 | 1 | 1 | 2 | 2 | 23 | M |
| *C. nasus* | 3 | 1 | 1 | 3 | 2 | 3 | 2 | 3 | 2 | 1 | 2 | 1 | 24 | H |
| *C. sardinella* | 3 | 1 | 1 | 3 | 2 | 2 | 2 | 3 | 2 | 1 | 2 | 1 | 23 | M |
| *Oncorhynchus clarki clarki* | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 3 | 1 | 1 | 2 | 27 | H |
| *O. clarki lewisi* | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 3 | 1 | 1 | 2 | 26 | H |
| *O. gorbuscha* | 3 | 2 | 1 | 3 | 2 | 1 | 1 | 2 | 2 | 1 | 2 | 1 | 21 | M |
| *O. keta* | 3 | 2 | 1 | 3 | 2 | 2 | 2 | 2 | 3 | 1 | 2 | 1 | 24 | H |
| *O. kisutch* | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 3 | 1 | 2 | 2 | 28 | H |
| *O. mykiss* | 3 | 3 | 2 | 3 | 2 | 2 | 2 | 2 | 2 | 1 | 1 | 2 | 25 | H |
| *O. mykiss* - steelhead | 3 | 3 | 1 | 3 | 2 | 2 | 2 | 2 | 3 | 1 | 1 | 2 | 25 | H |
| *O. nerka* | 3 | 2 | 1 | 3 | 2 | 2 | 2 | 2 | 3 | 1 | 2 | 1 | 24 | H |
| *O. nerka* - *kokanee* | 3 | 2 | 1 | 3 | 2 | 2 | 2 | 2 | 2 | 1 | 2 | 2 | 24 | H |
| *O. tshawytscha* | 3 | 2 | 2 | 3 | 2 | 3 | 2 | 2 | 3 | 1 | 2 | 2 | 27 | H |
| *Prosopium coulteri* | 3 | 1 | 1 | 3 | 2 | 1 | 2 | 3 | 1 | 1 | 2 | 1 | 21 | M |
| *P. cylindraceum* | 3 | 1 | 1 | 3 | 2 | 3 | 2 | 3 | 1 | 1 | 2 | 1 | 23 | M |
| *P. sp.* - giant pygmy whitefish | 3 | 1 | 1 | 3 | 2 | 2 | 2 | 3 | 1 | 1 | 2 | 1 | 22 | M |
| *P. williamsoni* | 3 | 2 | 2 | 3 | 2 | 2 | 2 | 3 | 3 | 1 | 2 | 2 | 27 | H |
| *Salmo salar* * | 3 | 2 | 2 | 3 | 2 | 2 | 2 | 2 | 3 | 1 | 2 | 1 | 25 | H |
| *Salmo trutta* * | 3 | 3 | 2 | 3 | 2 | 2 | 2 | 2 | 2 | 1 | 2 | 2 | 26 | H |
| *Salvelinus confluentus* | 3 | 3 | 2 | 3 | 2 | 3 | 2 | 2 | 3 | 1 | 2 | 2 | 28 | H |
| *Salvelinus fontinalis* * | 3 | 3 | 2 | 3 | 2 | 2 | 2 | 2 | 2 | 1 | 2 | 2 | 26 | H |
| *Salvelinus malma* | 3 | 3 | 3 | 3 | 2 | 3 | 2 | 2 | 3 | 1 | 2 | 2 | 29 | H |
| *Salvelinus namaycush* | 3 | 1 | 1 | 3 | 2 | 3 | 2 | 3 | 1 | 1 | 2 | 2 | 24 | H |
| *Stenodus leuchichthys* | 3 | 1 | 1 | 3 | 2 | 3 | 2 | 3 | 3 | 1 | 2 | 1 | 25 | H |
| *Thymallus arcticus* | 3 | 3 | 2 | 3 | 2 | 2 | 2 | 3 | 3 | 1 | 1 | 1 | 26 | H |

* indicates a species that has been introduced and become established in BC freshwaters

moderate sensitivity category. Most cyprind taxa (57%, 12/21) scored for moderate sensitivity; all other cyprinids (43%, 9/21) were classified as being of low sensitivity. British Columbia's two percid species (yellow perch and walleye) scored within the low and moderate sensitivity categories respectively. Three of BC's six gasterostids (50%) clustered within the moderate sensitivity category, while other taxa within this family all scored for lowest sensitivity. All four centrarchid species and both icturalid species (brown and black bullhead) scored within the lowest sensitivity grouping, as did the sole representatives of the esocid and hiodontid families (northern pike and goldeye respectively). In total, five of BC's 16 freshwater fish families (31%) had representatives within the highest sensitivity category, whereas four families (25%) were represented by species scoring only for lowest sensitivity. Of the fifteen introduced species established in BC waters (encompassing six families) eleven (73%) were classified as being of lowest sensitivity to logging impacts. Only three introduced species (all within the salmonid family) scored for highest sensitivity to logging impacts.

## DISCUSSION

All organisms display a wide range of adaptations (functional traits), related to resource acquisition, survival in the face of various threats, and reproduction. The quantitative analysis undertaken here suggests that functional traits displayed by a large proportion of British Columbia's freshwater fish taxa (encompassing at least seven families) makes them highly sensitive to habitat change associated with timber harvest. Our scoring and clustering approach, however, provides only a relative metric of sensitivity and validation of the results is dependent on existing and continuing field assessments of fish response to logging impacts. Our high sensitivity scoring for many of the salmonids, for example, is consistent with a wealth of studies that have documented significant long-term effects on various salmonid populations post-logging in the Pacific northwest (for examples see summaries in Meehan 1991 and Murphy 1995). Additionally, our analysis successfully clustered seven of nine BC fish taxa earlier identified as being particularly susceptible to logging in an independent province-wide review of land use impacts (Haas 1998). The other taxa previously identified that were not categorized as highly sensitive in this regard did, however, receive high scoring within our moderate sensitivity classification. A further fifteen taxa identified in G. Haas's (1998) document as having special forestry concerns were also successfully categorized by our analysis as being of either high or moderate sensitivity to logging. Conversely, our assessment that many of BC's introduced fish species are likely tolerant of logging impacts is consistent with a recent study by Waite and Carpenter (2000) that documented the prevalence of many of these same introduced species in impacted Oregon streams. They suggested that a reduction in riparian quality along with associated increases in temperature, nutrient, and sediment supply was the principal factor causing an apparent shift in fish assemblages from those dominated by native, intolerant species to those dominated by tolerant, introduced species. Further verification of our sensitivity rankings is dependent on comprehensive assessments of fish community characteristics pre-and-post logging, with an emphasis on determining response of lesser studied non-salmonid species. Given the absence of such information for most fish taxa in the province the use of detailed species trait information, combined with knowledge of environmental conditions, should provide our best method for predicting community response across a landscape. The principal value of this mechanistic approach is that relevant components (e.g., tolerances and selectivities) can be easily identified and understood, and they provide the biological basis for expectations of species response to change (Poff 1997).

There are, however, potential weaknesses with an analysis that relies on comparative aspects of species life history and ecology using information from widely disparate studies. For example, apparent differences in fish species response to physiochemical stressors may be misleading because variations in acclimation or experimental conditions can markedly affect results (Mathews 1987). There is also the potential for significant life history variation within species that may not be sufficiently captured even within our broad categorizations of traits. Basic information relating to life history, ecology and tolerances for many species of fish in British Columbia is lacking or is sketchy at best. For example, few studies specifically describe migratory behaviour in non-game stream fishes (Ross et al. 1985). Scott and Crossman's (1973) survey of Canadian freshwater fishes mentions migration for only 20% of the listed cyprinids; the remaining 80% have simply not been examined for migratory abilities (Smith 1991). The limited radio-tracking work done on cyprinids (e.g., Tyus 1985) suggests that some species may be capable of extensive seasonal migrations. Consequently, an acceptance of limited movement for many non-game species may be reflective simply of inadequate data rather than biological reality. For such poorly assessed traits we have had to rely on generalized familial descriptions and expert opinion for final scoring of our species, so that our assessments in such cases represents only a best guess. We would suggest that there exists a real and current need for basic studies on the life history and ecology of much of British Columbia's ichthyofauna. This would not only aid our present efforts to quantify species traits but is required to increase our understanding of general species-environment relationships.

Our ability to associate a specific biological response to a specific chemical or physical factor is also made difficult by the often interrelated and perhaps synergistic nature of causative factors (Rose 2000; Waite and Carpenter 2000). For example, we treated species traits in our scoring matrix as being independent of one another, whereas they are likely correlated to various degrees. Although we did not address this, understanding these correlations would be very useful as they could help define possible "syndromes" of traits (Poff 1997) that affect species sensitivity. Our choice of species traits for analysis also probably does not capture the full range of risk factors associated with logging activities. For example, riparian canopies are important in reducing levels of solar ultraviolet-B (UVB) penetration to lakes and streams. UV-B radiation has been shown to have negative impacts on aquatic productivity in general (Bothwell et al. 1994), as well as direct physiological effects on fish ranging from sunburn (Blazer et al. 1997) to suppression of immune function and egg mortality (Williamson et al. 1997; Salo et al. 1998). Potentially harmful intensities of UV-B radiation can reach depths of several meters in clear freshwater lakes and streams (Schindler et al. 1996). Preliminary analysis has suggested that solar UV radiation could be a factor in documented declines of salmonids in small streams (Walters and Ward 1998). It could therefore be very useful to score species for sensitivity for UV-B within our matrix. However, given that only limited comparative work on fish response to UV-B has been undertaken we did not feel justified in making species assessments at this time. Our analysis of species sensitivity was additionally restricted only to assessment of abiotic factors. Biotic interactions, although generally difficult to quantify (Poff 1997), could also be important in defining the response of fish species to logging impacts. For example, removal of forest canopy can induce changes in fish communities through temperature mediated effects on competitive ability (Hiram et al. 1987). In this regard Reeves et al. (1987) found that as temperatures in logged Oregon streams increased beyond 19°C redside shiners began to displace juvenile steelhead trout through exploitation competition for food resources, whereas at lower temperatures juvenile steelhead were dominant.

Despite these shortcomings we believe that quantifying species sensitivity represents an important step towards understanding fish response to habitat changes associated with logging or other land use impacts. Continued work on fish sensitivities might eventually allow us to

examine the effects of within-species variation in life history strategies or habitat use, perhaps permitting "customized" predictions of response by individual populations of fish. Ultimately, however, we envision an assessment of sensitivity as only one element in fully defining and quantifying logging risk to fish species in the province. Species rarity and land use represent two other components that need to be incorporated into a complete analysis of current and future risk to fish. Small populations with limited distributions are, by their nature, more vulnerable to natural accidents and human-caused alterations of habitat. Consequently, rare species will generally have greater attendant risks of genetic impoverishment or extinctions regardless of inherent sensitivity (Sheldon 1987), particularly if rare populations coincide with areas of intensive development. Not surprisingly, the majority of documented fish extinctions in North America have involved species with restricted ranges (Miller et al. 1989). Having quantified species sensitivity as outlined in this paper the challenge then becomes how to quantify species rarity and the overlap of species distributions with negative land use impacts.

BC's Watershed Atlas GIS system (BC MoELP 1996) provides a spatial template for quantifying species rarity. The Watershed Atlas has defined the boundaries of approximately 18,000 watersheds for British Columbia, which can be overlain with known and modeled fish distributions. Rarity for each of the province's fish species can then be scored based on the percentage of the province's watersheds known to contain a particular species based on historical survey records from government and university archives. For watersheds with no fish collection data, species presence can be modeled based on large-scale habitat and climate information derived from the Watershed Atlas and other government data sets (e.g., Environment Canada 1994). Earlier work on modeling fish species distributions in the province (e.g., Porter et al. 1998; Porter et.al. 2000; Rosenfeld et al. in review) has relied on directed fish sampling undertaken within defined watershed confines. Development of provincial fish models employing archived (historical) fish collection data can present greater problems associated with data reliability, variation in collection techniques and sampling periods, or differential taxonomic resolution and accuracy across sites (Poff and Allan 1995). An analysis of recent BC Fisheries inventories (Haas et al. submitted) detected errors in fish species identifications averaging over 26%, with substantial mistakes also being made at the genera and family levels. Consequently, species distribution models should be based only on survey records either confirmed by experts or by repeated and independent sampling of the same species within a watershed.

The Watershed Atlas can also be used to determine a score for the overlap of species distributions and land use impacts, based on the percentage of a species range that has been historically logged and the area anticipated to be logged within the next five years (i.e., normal planning cycles of the provincial forestry ministry). Environmental statistics relating to provincial land use are regularly summarized by the BC government and have recently been converted into formats compatible with the BC Watershed Atlas (BC MoELP 1999) and can therefore be overlain with fish distributions. A number of studies have shown that roads serve as the best general proxy of cumulative effects associated with land use and human access (see Trombulak and Frissell 2000). The extent and intensity of past logging activity can be inferred from forest age information and from calculation of road densities within the province's watersheds, as historical timber harvest was heavily reliant on road construction. In addition to the more direct impacts of logging on fish populations associated road building can allow increased human access into formerly remote areas, followed typically by increases in both legal and illegal fishing in streams and lakes. Native fish in pristine areas are often vulnerable to even small increases in fishing effort, and public pressure for greater stocking in these areas often comes at the expense of resident species or populations (Trombulak and Frissell 2000). The extent of future logging activities will be determined from GIS overlays of the BC Ministry of Forest's regional 5-year plans for timber harvest in the province. Digital maps delineating these 5

year plans are not currently available but are part of ongoing development within the Forestry Ministry's regional offices (D. Morgan, Planning Systems Biologist, BC Ministry of Forests, pers. comm.).

A final scoring of fish species for logging risk would then be based on three independent components, such that:

$$\text{Logging Risk} = \text{Sensitivity} + \text{Rarity} + \text{Land Use}$$

According to this proposed algorithm, species perceived as being at greatest risk would be: 1) highly sensitive to logging impacts, 2) have limited range (high rarity score), and 3) have a large proportion of their existing range both historically logged and scheduled to be further logged over the next 5 years. Species perceived as being at lowest risk would be: 1) relatively insensitive to logging impacts, 2) have a broad distribution (low rarity score), and 3) have only a minimal percentage of their existing range either historically logged or slated to be logged in the next 5 years. A comparative ranking of the province's fish species for perceived risk could be established based on the composite score produced using the three algorithm components. The transparency of this simple model should make it both easy to refine and temporally dynamic. For example, if improved species trait information became available the sensitivity matrix for a species could be quickly rescored and the risk ranking recalculated using the already existing rarity and land use scores. Similarly, updated survey information could be used to adjust species distribution models and recalculate rarity scores. Finally, the land use scores could be changed regularly as continued implementation of logging plans is likely to shift development impacts into different areas of the province.

Understanding the effects of timber harvest and other anthropogenic changes on fish populations is critical for effective resource management. However, modeling these effects through traditional deterministic approaches has proven difficult as fish often show significant variations in life histories, may utilize a variety of habitats over their lifetimes, and exhibit large natural fluctuations in abundance (Rose 2000). Poff and Allan (1995) have suggested that detecting fish response to broad-scale landscape changes could be facilitated by use of functional species traits, which can be generalized across broad geographic domains. By focusing on species traits and limiting environmental factors we may be able to increase the ability to predict response to habitat change, solely on the basis of simple mechanistic relationships. In this paper we sought to link key functional species traits to environmental changes associated with logging activities, and to establish a method of scoring BC fish species for sensitivity to these changes. Our efforts suggest that greater research would help in more precisely defining the life histories, ecology and environmental tolerances of many of these species. Although we have been limited to broad categorizations of functional traits we believe, however, that the proposed framework will contribute to improved quantification and prediction of species sensitivity. Scoring species for sensitivity, in conjunction with species distribution models developed using geo-referenced survey data and GIS overlays of provincial land use patterns, could represent a novel approach for assessing the relative risk of British Columbia's fish fauna to the continuing effects of timber harvest.

# REFERENCES


Abbe, T.B., and D.R. Montgomery. 1996. Large river debris jams, channel hydraulics and habitat formation in large rivers. Regulated Rivers Research & Management 12: 201-221.

Anderson, P.A., B.R. Taylor, and G. Balch. 1996. Quantifying the effects of sediment release on fish and their habitats. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2346.

Balon, E.K. 1975. Reproductive guilds of fishes: a proposal and definition. Journal of the Fisheries Research Board of Canada 32: 821-864.

Barton, B.A., and B.R. Taylor. 1994. Dissolved oxygen requirements for fish of the Peace, Athabasca and Slave River basins. Northern Rivers Basins Study Project Report No. 29. Northern River Basins Study, Edmonton, Alberta.

BC MoELP (British Columbia Ministry of Environment, Lands and Parks). 1996. An introduction to the British Columbia Watershed Atlas. Victoria, BC.

BC MoELP. 1999. Watersheds BC user's guide: environmental statistics (draft). British Columbia Ministry of Environment, Lands and Parks. Geographic Data BC. Victoria, BC.

Beamish, R.J. 1987. Status of the Lake Lamprey, *Lampetra macrostoma*, in Canada. Canadian Field-Naturalist 101(2): 186-189.

Berkman, H.E., and C.F. Rabeni. 1987. Effect of siltation on stream fish communities. Environmental Biology of Fishes 18(4): 285-294.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research 14: 1011-1016.

Beschta, R.L., and W.L. Jackson. 1979. The intrusion of fine sediments into a stable gravel bed. Journal of the Fisheries Research Board of Canada 36: 204-210.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. Pages 191-232 *in* E.O Salo and T.W. Cundy (Editors). Streamside management: forestry and fishery interactions. University of Washington, Institute of Forest Resources Contribution No. 57, University of Washington, Seattle.

Bilby, R.E., and J.W. Ward. 1989. Changes in characteristics and function of woody debris with increasing size of streams in western Washington. Transactions of the American Fisheries Society 118: 368-378.

Bilby, R.E., B.R. Fransen, and P.A. Bisson. 1996. Incorporation of nitrogen and carbon from spawning coho salmon into the trophic system of small streams: evidence from stable isotopes. Canadian Journal of Fisheries and Aquatic Sciences 53:164-173.

Bisson, P.A., R.E. Bilby, M.D. Bryant, C.A. Dolloff, G.B. Grette, R.A. House, M.L. Murphy, K.V.Koski, and J.R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. Pages 143-190 *in* Salo, E.O., and T.W. Cundy (Editors).

Streamside management: forestry and fishery interactions. University of Washington, Institute of Forest Resources Contribution No. 57, University of Washington, Seattle.

Black, E.C. 1953. Upper lethal temperatures for some British Columbia fishes. Journal of the Fisheries Research Board of Canada 10(4): 196-210.

Blazer, V.S., D.L. Fabacher, E.E. Little, M.S. Ewing, and K.M. Kocan. 1997. Effects of ultrviolet-B radiation on fish: histological comparison of a UVB-sensitive and UVB tolelerant species. Aquatic Animal Health 9: 132-143.

Bothwell, M.L., D.M.J. Sherbot, C.M. Pollock. 1994. Ecosystem response to solar ultraviolet-B radiation: influence of trophic-level interactions. Science (Washington) 265: 97-100.

Bjornn, T.C., and D.W. Reiser. 1991. Habitat requirements of salmonids in streams. Pages 83-138 *in* W.R. Meehan (editor). Influences of forest and rangeland management on salmonid fishes and their habitats. Special publication 19, American Fisheries Society, Bethesda, MD.

Camargo, J.A. 1991. Ecotoxicological analysis of the influence of industrial effluent on fish populations in a regulated stream. Aquaculture and Fisheries Management 22: 509-518.

Cannings, S.G., and J.Ptolemy. 1998. Rare freshwater fish of British Columbia. Province of British Columbia, BC Ministry of Evironment, Lands and Parks. Conservation Data Centre. Victoria, BC

Cederholm, C.J., and L.M. Reid. 1987. Impact of forest management on coho salmon (*Oncorhynchus kisutch*) populations of the Clearwater River, Washington: a project summary. Pages 373-398 *in* Salo, E.O., and T.W. Cundy (Editors). Streamside management: forestry and fishery interactions. University of Washington, Institute of Forest Resources Contribution No. 57, University of Washington, Seattle.

Chamberlin, T.W., R.D. Harr, and F.H. Everest. 1991. Timber harvesting, silviculture, and watershed process. Pages 181-205 *in* W.R. Meehan (editor). Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19. American Fisheries Society, Bethesda, MD.

Chapman, D.W., and E. Knudson. 1980. Channelization and livestock impacts on salmonid habitat and biomass in western Washington. Transactions of the American Fisheries Society 109:357-363.

Coutant, C.C. 1977. Compilation of temperature preference data. Journal of Fisheries Research Board of Canada 34: 739-745.

Crecco, V.A., and T.F. Savoy. 1984. Effects of fluctuations in hydrographic conditions on year-class strength of American shad (*Alosa sapidissima*) in the Connecticut River. Canadian Journal of Fisheries and Aquatic Sciences 41:1216-1233.

Cunjak, R.A. 1996. Winter habitat of selected stream fishes and potential impacts from land-use activity. Canadian Journal of Fisheries and Aquatic Sciences 53(Suppl. 1): 267-282.

Dauble, D.D. 1986. Life history and ecology of the largescale sucker (*Catostomus macrocheilus*) in the Columbia River. The American Midland Naturalist 116(2): 356-367.

Davis, J.C. 1975. Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review. Journal of the Fisheries Research Board of Canada 32(12): 2295-2332.

Detenbeck, N.E., and P.W. DeVore, G.J. Niemi, and A. Lima. 1992. Recovery of temperate-stream fish communities from disturbance: a review of case studies and synthesis of theory. Environmental Management 16: 33-53.

Dolloff, C.A. 1986. Effects of stream cleaning on juvenile coho salmon and Dolly Varden in southeast Alaska. Transactions of the American Fisheries Society 115: 743-755.

Doudoroff, P., and D.L. Shumway. 1970. Dissolved oxygen requirements of freshwater fishes. FAO Fisheries Technical Paper No. 86. Food and Agriculture Organization of the United Nations, Rome.

Dryfoos, R.L. 1965. The life-history and ecology of the longfin smelt in Lake Washington. PhD thesis, University of Washington, Seattle, WA, USA.

Dunham, J.B., and B.E. Rieman. 1999. Metapopulation structure of bull trout: influences of physical, biotic, and geometrical landscape characteristics. Ecological Applications 9(2): 642-655.

Eaton, J.G., J.H. McCormick, B.E. Goodno, D.G. O'Brien, .G. Stefany, M. Hondzo, and R.M. Scheller. 1995. A field information-based system for estimating fish temperature tolerances. Fisheries 20(4); 10-18.

Elliot, S.T. 1986. Reduction of a Dolly Varden population and macrobenthos after removal of logging debris. Transactions of the American Fisheries Society 115: 392-347.

Environment Canada. 1994. Canadian climate normals 1961-1990. CMCD softwate version 3.0.

Erman, D.C., E.D. Andrews, and M. Yoder-Williams. 1988. Effects of winter floods on fishes in the Sierra Nevada. Canadian Journal of Fisheries and Aquatic Sciences 45: 2195-2200.

Fausch, K.D., and M.K. Young. 1995. Evolutionary significant units and movement of resident stream fishes: a cautionary tale. Pages 360-370 *in* J.L. Nielson, editor. Evolution and the aquatic system: defining unique units in population conservation. American Fisheries Society, Symposium 17, Bethesda, Maryland.

Ford, B.S., P.S. Higgins, A.F. Lewis, K.L. Cooper, T.A. Watson, G.L.Ennis, and R.L. Sweeting. 1992. Literature reviews of the life history, habitat requirements and mitigation/compensation strategies for 13 species of sport fish in the Peace and Columbia River drainages of British Columbia. Canadian Department of Fisheries and Oceans Report. Vancouver, BC, Canada.

France, R. 1997. Land-water linkages: influences of riparian deforestation on lake thermocline depth and possible consequences for cold stenotherms. Canadian Journal of Fisheries and Aquatic Sciences. 54(6): 1299-1308.

Fuller, W.A. 1955. The inconnu (*Stenodus leuchichthys*) in Great Slave Lake and adjoining waters. Journal of the Fisheries Research Board of Canada 12(50: 768-780.

Furniss, M.J., T.D. Roelefs, and C.S.Yee. 1991. Road construction and maintenance. Pages 297-323 *in* W.R. Meehan (editor). Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19. American Fisheries Society, Bethesda, MD.

Gardner, M.B. 1981. Effects of turbidity on feeding rates and selectivity of bluegills. Transactions of the American Fisheries Society 110:446-450.

Gibbons, D.R., W.R. Meehan, K.V. Koski, and T.R. Merrell, Jr. 1987. History of fisheries and forestry interactions in southeastern Alaska. Pages 297-329 *in* E.O Salo and T.W. Cundy (Editors). Streamside management: forestry and fishery interactions. University of Washington, Institute of Forest Resources Contribution No. 57, University of Washington, Seattle.

Gray, L.J. 1981. Species composition and life histories of aquatic insects in a lowland Sonoran Desert stream. American Midland Naturalist 106: 229-242.

Gregory, R.S. 1993. Effect of turbidity on the predator avoidance behaviour of juvenile chinook salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 50:241-246.

Gregory, R.S., and C.D. Levings. 1998. Turbidity reduces predation on migrating juvenile pacific salmon. Transactions of the North American Fisheries Society 127: 275-285.

Groot, C., and L. Margolis. (Editors). 1991. Pacific salmon life histories. Canada Department of Fisheries and Oceans, Vancouver, BC.

Haas, G.R. 1998. Indigenous fish species at risk in BC, with recommendations and prioritizations for conservation, forestry/resource use, inventory and research. Province of British Columbia, Ministry of Fisheries, Fisheries Management Report No. 105, Vancouver, BC.

Haas, G.R., J.D. McPhail, and L.M. Ritchie. [submitted]. Species identification errors, problems, and their consequences: the value and lost skills of basic natural history and taxonomy. [available from GRH at BC Fisheries Research Branch, UBC, 2204 Main Mall, Vancouver, BC, V6T 1Z4].

Hall, J.D., and R.L. Lantz. 1969. Effects of logging on the habitat of coho salmon and cutthroat trout in coastal streams. Pages 355-375 *in* T.G. Northcote (Editor). Symposium on salmon and trout in streams. H.R. MacMillan Lectures in Fisheries, University of British Columbia, Institute of Fisheries, Vancouver.

Hallock, R.J., R.F. Elwell, and D.H. Fry, Jr. 1970. Migrations of adult king salmon *Oncorhynchus tshawytscha* in the San Joaquin Delta as demonstrated by the use of sonic tags. California Department of Fish and Game. Fish Bulletin 151.

Harr, R.D., and R.L. Fredrickson, 1979. Some characteristics and consequences of snowmelt during rainfall in western Oregon. Forest Sciences Laboratory, Corvallis, OR. Pacific Northwest Forest and Range Experiment Station Report.

Harr, R.D., R.L. Fredrickson, and J.S. Rothacher. 1979. Changes in stream flow following timber harvest in southwestern Oregon. Research Paper PNW-249. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, OR.

Harrison, G.W. 1979. Stability under environmental stress: resistance, resilience, persistence, and variability. American Naturalist 113: 659-669.

Hartman, G.F. and J.C. Scrivener. 1990. Impacts of forest practices on a coastal stream ecosystem, Carnation Creek, British Columbia. Canadian Bulletin of Fisheries and Aquatic Sciences 223.

Harvey, BC 1987. Susceptibility of young-of-the-year fishes to downstream displacement by flooding. Transactions of the North American Fisheries Society 116: 851-855.

Hauser, W.J. 1969. Life-history of the mountain sucker, *Castostomus platyrhynchos*, in Montana. Transactions of the American Fisheries Society 98: 209-215.

Hawkins, C.P., and J.R. Sedell. 1981. Longitudinal and seasonal changes in functional organization of macroinvertebrate communities in four Oregon streams. Ecology 62:387-397.

Hawkins, C.P, M.L. Murphy, and N.H. Anderson. 1982. Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology 63:1840-1856.

Hibbert, A.R. 1967. Forest treatment effects on water yield. Pages 527-543 *in* Sopper, W.E., and H.W. Lull (Editors). Forest hydrology: proceedings, forest hydrology seminar. Pergamon Press, New York.

Hicks, B.J., J.D. Hall, P.A. Bisson, and J.R. Sedell. 1991. Response of salmonids to habitat change. Pages 483-518 *in* W.R. Meehan (editor). Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19. American Fisheries Society, Bethesda, MD.

Hicks, B.J., R.L. Beschta, and R.D. Harr. 1991. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. Water Resources Bulletin: 27:217-226.

Hiram, W.L., C.B. Schreck, C.E. Bond, and E. Rexstad. 1987. Factors influencing changes in fish assemblages of Pacific northwest streams. *In:* Mathews, W.J., and D.C. Heins. Community and evolutionary ecology of North American stream fishes. University of Oklahama Press: Norman and London.

Hogan, D.L. 1986. Channel morphology of unlogged, logged, and debris torrented streams in the Queen Charlotte Islands. British Columbia Ministry of Forests Land Management Report No 49.

Hokanson, K.E.F. 1977. Temperature requirements of some percids and adaptations to the seasonal temperature cycle. Journal of the Fisheries Research Board of Canada 43: 1524-1550.

Hughes, G.W., and A.E. Peden. 1989. Status of the Umatilla Dace, Rhinichthys umatilla, in Canada. Pages 193-200. *in* R.R. Campbell (ed.). Rare and endangered fishes and marine mammals of Canada: Status Reports V. The Canadian Field-Naturalist 103(2): 147-220.

Hughes, R.M., P.R. Kaufmann, A.T. Herlihy, T.M. Kincaid, L. Reynolds, and D.P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. Canadian Journal of Fisheries Management 55: 1618-1631.

Ice, G.W. 1985. Catalogue of landslide inventories for the northwest. National Council of the Paper Industry for Air and Stream Improvement, Technical Bulletin 456, New York.

Johnson, J.E., and R.T. Hines. 1999. Effect of suspended sediment on vulnerability of young razorback suckers to predation. Transactions of the American Fisheries Society 128: 648-655.

Johnston, C.E. 1999. The relationship of spawning mode to conservation of North American minnows (Cyprinidae). Environmental Biology of Fishes 55: 21-30.

Johnston, N. T., and P.A. Slaney. 1996. Fish Habitat Assessment Procedures. Province of British Columbia, Ministry of Environment, Lands and Parks, and Ministry of Forests. Watershed Restoration Technical Circular No 8.

Jones, J.A., and G.E. Grant. 1996. Cumulative effects of forest harvest on peak streamflow in the western Cascades of Oregon. Water Resources Research 32: 959-974.

Jones, J.A., F.J. Swanson, Wemple, BC, and K.U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14(1): 76-85.

Lane, E.D. 1991. Status of the White Sturgeon, *Acipenser transmontanus*, in Canada. Canadian Field-Naturalist 105(2): 161-168.

Larkin, G.A., and P.A. Slaney. 1996. Calibration of a habitat sedimentation indicator for use in measuring the effectiveness of watershed restoration treatments. Province of British Columbia, Ministry of Environment, Lands and Parks and Ministry of Forests, Watershed Restoration Management Report No.5.

Larkin, G.A., and P.A. Slaney. 1997. Implications of trends in marine-derived nutrient influx to south coastal salmonid production. Fisheries 22(11): 16-24.

Larkin, G.A., P.A. Slaney, P. Warburton, and A.S. Wilson. 1998. Suspended sediment and fish habitat sedimentation in central interior watersheds of British Columbia. Province of British Columbia, Ministry of Environment, Lands and Parks and Ministry of Forests, Watershed Restoration Management Report No.7.

Lehtinen, R.M., N.D. Mundahl, and J.C. Madejczyk. 1997. Autumn use of woody snags by fish in backwater and channel border habitats of a large river. Environmental Biology of Fishes 49: 7-19.

Lloyd, D.S., J.P. Koenings, and J.D. LaPierre. 1987. Effects of turbidity in fresh waters of Alaska. North American Journal of Fisheries Management 7:18-33.

Mathews, W.J. 1987. Physiochemical tolerance and selectivity of stream fishes as related to their geographic ranges and local distributions. In Mathew, W.J., and D.C. Heins. Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press: Norman and London.

McLeay, D.J., Birtwell, I.K., G.F. Hartman, and G.L. Ennis. 1987. Responses of Arctic grayling, *Thymallus arcticus*, to acute and prolonged exposure to Yukon placer mining sediment. Canadian Journal of Fisheries and Aquatic Sciences 44: 658-673.

McNeil, W.J. 1964. Effect of the spawning bed environment on reproduction of pink and chum salmon. Fishery Bulletin 65: 495-523.

McPhail, J.D. 1987. Status of the Salish Sucker, *Catostomus sp.*, in Canada. Canadian Field-Naturalist 101(2): 231-236.

McPhail, J.D., and J.S. Baxter. 1996. A review of bull trout (*Salvelinus confluentus*) life-history and habitat use in relation to compensation and improvement opportunities. Province of British Columbia, Ministry of Environment, Lands and Parks, Fisheries Management Report No. 104.

Meehan, W.R. 1974. The forest ecosystem of southeast Alaska; 3. Fish habitats. U.S. Forest Service General Technical Report PNW-15.

Meehan, W.R. (editor). 1991. Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19. American Fisheries Society, Bethesda, MD.

Meisner, J.D., J.S. Rosenfeld and H.A. Regier. 1988. The role of groundwater in the impact of climate warming on stream salmonines. Fisheries 13(3): 2- 8.

Milligan, G.W. 1980. An examination of the effect of six types of error perturbation on fifteen clustering algorithms. Psychometrika 45: 325-342.

Miller, R.R., J.D. Williams, and J.E. Williams. 1989. Extinctions of North American fishes during the past century. Fisheries 14: 22-38.

Miller, L.B., D.J. McQueen, and L. Chapman. 1997. Impacts of forest harvesting on lake ecosystems: a preliminary literature review. Province of British Columbia, Ministry of Environment, Lands and Parks, Wildlife Branch, Wildlife Bulletin No. B-84. Victoria, BC.

Moodie, G.E. 1984. Status of the Giant (Mayer Lake) Stickleback, *Gasterosteus sp.*, on the Queen Charlotte Islands, British Columbia. Canadian Field-Naturalist 98(1): 115-119.

Monan, G.E., J.H. Johnson, and G.F. Esterberg. 1975. Electronic tags and related tracking techniques aid in study of migrating salmon and steelhead trout in the Columbia River basin. Marine Fisheries Review 37(2):9-15.

Montgomery, D.R., E.R. Beamer, G.R.Pess, and T.P. Quinn. 1999. Channel type and salmonid spawning distribution and abundance. Canadian Journal of Fisheries and Aquatic Sciences 56: 377-387.

Murphy, M.L., and J.D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Canadian Journal of Fisheries and Aquatic Sciences 38:137-145.

Murphy, M.L., J. Heifetz, S.W. Johnson, K.V. Koski, and J.F. Thedinga. 1986. Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Canadian Journal of Fisheries and Aquatic Sciences 43:1521-1533.

Murphy, M.L., and K.V. Koski. 1989. Input and depletion of woody debris in Alaska streams and implications for streamside management. North American Journal of Fisheries Management 9:427-436.

Murphy, M.L. and W.R. Meehan. 1991. Stream ecosystems. Pages 17-46 *in* W.R. Meehan (editor). Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19. American Fisheries Society, Bethesda, MD.

Murphy, M.L. 1995. Forestry impacts on freshwater habitat of anadromous salmonids in the Pacific Northwest and Alaska -- requirements for protection and restoration. U.S. Department of Commerce (NOAA) Coastal Ocean Program. Decision Analysis Series No. 7.

Newcombe, C.P. 1985. Harmful effects of suspended sediments to trout, salmon and other organisms in aquatic ecosystems, as a function of increasing concentration and duration of exposure: annotations and bibliography. Province of British Columbia, Ministry of Environment, Lands and Parks, Fisheries Branch, Victoria, BC

Newcombe, C.P., and D.D. McDonald. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11: 63-72.

Newcombe, C.P. and J.O Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4): 693-727.

Northcote, T.G. 1997. Potamodromy in Salmonidae - living and moving in the fast lane. North American Journal of Fisheries Management 17: 1029-1045.

Orians, G. 1980. Micro and macro in ecological theory. Bioscience 30:79.

Parent, S. and L.M. Schrimi. 1995. A model for the determination of fish species at risk based upon life-history traits and ecological data. Canadian Journal of Fisheries and Aquatic Sciences 52: 1768-1781.

Pearson, M. 1998. A review of the distribution, status and biology of the endangered Salish Sucker (*Catastomus sp.*) and Nooksack Dace (*Rhinichthys sp.*). Province of British Columbia, Ministry of Fisheries, Fisheries Technical Circular No. 101.

Peden, A.E., and G.W. Hughes. 1984. Status of the Speckled Dace, *Rhinichthys osculus*, in Canada. Canadian Field-Naturalist 98(1): 98-103.

Pianka, E.R. 1970. On r- and K- selection. American Naturalist 104: 592-597.

Poff, N.L. 1997. Landscape filters and species traits; towards a mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 16(2): 391-409.

Poff, N.L, and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76: 606-627.

Porter, M., J. Rosenfeld, and E. Parkinson. 1998. Macrohabitat use and predictive models of fish distribution in the Blackwater drainage. Province of British Columbia, Ministry of Fisheries. Fisheries Management Report No. 108.

Porter, M., J. Rosenfeld and E.Parkinson. Predictive models of fish species distribution in the Blackwater drainage, British Columbia. North American Journal of Fisheries Management 20: 349-359.

Rader, R.B. 1997. A functional classification of the drift: traits that influence invertebrate availability to salmonids. Canadian Journal of Fisheries and Aquatic Sciences 54.

Reeves, G.H., F.H. Everest, and T.E. Nickelson. 1987. Interactions between the redside shiner (*Richardsonius balteatus*) and the steelhead trout (*Salmo gairdneri*) in western Oregon: the influence of water temperature. Canadian Journal of Fisheries and Aquatic Sciences 44: 1603-1613.

Reid, L.M., and T.Dunne. 1984. Sediment production from forest road surfaces. Water Resources Research 20: 1753-1761.

Rood, K.M. 1984. An aerial photograph inventory of the frequency and yield of mass wasting on the Queen Charlotte Islands, British Columbia. British Columbia Ministry of Forests, Land Management Report 34.

Rose, K.A. 2000. Why are quantitative relationships between environmental quality and fish populations so elusive? Ecological Applications 10(2): 367-385.

Rosenfeld, J. M.S. Porter, E.A. Parkinson. 2000. Habitat factors affecting the abundance and distribution of juvenile cutthroat trout and coho salmon. Canadian Journal of Fisheries and Aquatic Sciences 57: 766-774.

Ross, S.T., W.J. Matthews, A.A. Echelles. 1985. Persistence of stream fish assemblages: effects of environmental change. The American Naturalist 126: 24-40.

Rothacher, J. 1971. Regimes of streamflow and their modification by logging. Pages 40-53 *in* Krygier, J.T., and J.D. Hall. 1971 (Editors). Proceedings, forest land uses and stream environment symposium. Continuing Education Publications, Oregon State University, Corvallis.

Salo, H.M., T.M. Aalton, S.E. Markkula, and E.I Jokinen. 1998. Ultraviolet B irradiation modulates the immune system of fish (*Rutilus rutilus*, Cyprinidae). Photochemistry and Photobiology 67: 433-437.

Schindler, D.W., P.J. Curtis, B.R. Parker, and M.P. Stainton. 1996. Consequences of climate warming and lake acidification for UV-B penetration in North American boreal lakes. Nature (London) 379: 705-708.

Schmidt, D., and W.J. O'Brien. 1982. Plankitovorous feeding ecology of arctic grayling (*Thymallus arcticus*). Canadian Journal of Fisheries and Aquatic Sciences 39: 475-482.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Fisheries Research Board of Canada Bulletin No. 184. Ottawa, Ontario.

Seegrist, D.W., and R.Gard. 1972. Effects of floods on trout in Sagehen Creek, California. Transactions of the American Fisheries Society 101: 478-482.

Servizi, J.A., ar ' D.W. Martens. 1991. Effect of temperature, season, and fish size on acute lethality of pended sediments to coho salmon, *Oncorhynchus kisutch*. Canadian Journal of Fisheries and Aquatic Sciences 48: 493-497.

Sheldon, A.L. 1987. Rarity: patterns and consequences for stream fishes. *In:* Mathews, W.J., and D.C. Heins. Community and evolutionary ecology of North American stream fishes. University of Oklahama Press: Norman and London.

Shepard, B.G., G.F. Hartman and W.J. Wilson. 1986. Relationships between stream and intragravel temperatures in coastal drainages, and some implications for fisheries workers. Canadian Journal of Fisheries and Aquatic Sciences 43: 1818- 822.

Shindler, D.W., P. J. Curtis, B.R. Parker, and M.P. Stainton. 1996. Consequences of climate warming and lake acidification for UV-B penetration in North American boreal lakes. Nature 379(22); 705-708.

Shortreed, K.S., and J.G. Stockner. 1983. Periphyton biomass and species composition in a coastal rainforest stream in British Columbia: effects of environmental changes caused by logging. Canadian Journal of Fisheries and Aquatic Sciences 40:1887-1895.

Sigler, J.W., T.C. Bjornn, and F.H. Everest. 1984. Effects of chronic turbidity on density and growth of steelhead and coho salmon. Transactions of the American Fisheries Society 113: 142-150.

Slaney, P.A., T.G. Halsey, and A.F. Tautz. 1977. Some effects of forest harvesting on salmonid rearing habitat in two streams in the central interior of British Columbia. Province of British Columbia, Ministry of Environment, Fisheries Management Branch, Fisheries Management Report No 71.

Slaney, P.A., and D. Zaldokas. 1997. Fish Habitat Rehabilitaton Procedures. Province of British Columbia, Ministry of Environment, Lands and Parks, Watershed Restoration Technical Circular No. 9.

Smale, M.A. and C.F. Rabeni. 1995a. Influences of hypoxia and hyperthermia on fish species composition in headwater streams. Transactions of the American Fisheries Society 124: 711-725.

Smale, M.A. and C.F. Rabeni. 1995b. Hypoxia and hyperthermia tolerances of headwater stream fishes. Transactions of the American Fisheries Society. 124: 698-710.

Smith, R.J.F. 1991. Social behaviour, homing and migration. *In:* Winfield, I.J. and J.S. Nelson (eds.). Cyprinid fishes: systematics, biology and exploitation. Chapman and Hall, London, UK.

Stockner, J.G., E. Rydin, and P. Hyenstrand. 2000. Cultural oligotrophication: causes and consequences for fisheries resources. Fisheries 25(5): 7-14.

Swanston, D.N. 1991. Natural processes. Pages 139-179 *in* W. R. Meehan (editor). Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19, American Fisheries Society, Bethesda, MD.

Townsend, C.R., and A.G. Hildrew. 1994. Species traits in relation to a habitat template for river systems. Freshwater Biology 31: 265-275.

Tripp, D.B., and V.A. Poulin. 1986. The effects of logging and mass wasting on salmonid spawning habitat in streams on the Queen Charlotte Islands. Land Management Report No. 50, British Columbia Ministry of Forests and Lands, Victoria, BC.

Trombulak, S.C., and C.A. Frissell. 2000. Review of the ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14(1): 18-30.

Tschaplinski, P.J., and G.F. Hartman. 1983. Winter distribution of juvenile coho salmon (*Oncorhynchus kistutch*) before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival. Canadian Journal of Fisheries and Aquatic Sciences 40: 452-461.

Tyus, H.M. 1985. Homing behaviour noted for Colorado squawfish. Copeia 1985(1): 213-215.

Volpe, J.P., E.B. Taylor, and B.W Glickman. 2000. Evidence of natural reproduction of aquaculture-escaped Atlantic salmon in a coastal British Columbia river. Conservation Biology 14(3): 899-903.

Waite, I.A., and K.D. Carpenter. 2000. Associations among fish assemblage structure and environmental variables in Willamette Basin streams, Oregon. Transactions of the American Fisheries Society 129: 754-770.

Walters, C., and B. Ward. 1998. Is solar radiation responsible for declines in marine survival rates of anadromous salmonids that rear in small streams? Canadian Journal of Fisheries and Aquatic Sciences 55: 2533-2538.

Warren, M.L., and M.G. Pardew. 1998. Road crossings as barriers to small-stream fish movement. Transactions of the American Fisheries Society 127: 637-644.

Wilkinson, L., L. Engelman, J. Corter and M. Coward. 1996. Cluster analysis; a supplementary module for SYSTAT. SYSTAT, Evanston, Ill.

Williamson, C.E., S.L. Metzger, P.A., Lovera, and R.E. Moeller. 1997. Solar ultraviolet radiation and the spawning habitat of yellow perch, *Perca flavescens*. Ecological Applications 7: 1017-1023.

Winfield, I.J., and J.S. Nelson (eds.). Cyprinid fishes: systematics, biology and exploitation. Chapman and Hall, London, UK.

Winn, H.E. 1960. Biology of the brook stickleback *Eucalia inconstans* (Kirtland). American Midland Naturalist 63(2): 424-438.

Winston, M.R., C.M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming a prairie stream. Transactions of the American Fisheries Society 120: 98-105.

Wooton, R.J. 1984. A functional biology of sticklebacks. Croom Helm Ltd. London, UK.

Appendix 1. Scoring for fish species life history traits with associated literature sources.

| Family/ Species name | Common name | Origin[a] | T | HS | HR | O2 | ST | RS | FM | SM | MS | WC | SS | C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Acipenseridae** | | | | | | | | | | | | | | |
| *Acipenser medirostris* | Green Sturgeon | N | 2[1] | 1[1] | 1[1] | 2[50,1] | 1[1] | 3[24,1] | 1[1] | 3[1] | 2[1] | 1[1] | 1[1] | 1[1] |
| *A. transmontanus* | White Sturgeon | N | 2[5,7,44,66] | 1[5,44,49] | 1[5,1,44] | 2[50,1] | 1[5,49] | 3[24,1] | 1[5,49] | 3[5,49] | 2[5,44,49] | 1[5,1] | 1[5,44,49] | 1[5] |
| **Clupeidae** | | | | | | | | | | | | | | |
| *Alosa sapidissima* | American Shad | I | 1[2] | 1[1] | 1[1] | 2[50,51] | 2[8] | 3[24,1] | 2[1,12] | 1[1] | 3[1] | 1[1] | 1[1] | 1[1] |
| **Centrarchidae** | | | | | | | | | | | | | | |
| *Lepomis gibbosus* | Pumpkinseed Sunfish | N | 1[52] | 1[1] | 1[1] | 2[50,66] | 2[64,66] | 2[24,1] | 2[1] | 1[1] | 1[1] | 1[1] | 1[1] | 1[1] |
| *Micropterus dolomieui* | Smallmouth Bass | I | 1[3,4] | 1[1] | 1[1] | 2[50,4,66] | 2[64,66] | 2[24,1] | 2[1] | 1[9,1] | 1[1] | 1[1] | 1[1] | 2[11,1] |
| *M. salmoides* | Largemouth Bass | I | 1[2,3,4] | 1[1] | 1[1] | 1[50,51,4,66] | 2[64,66] | 2[24,1] | 2[1] | 1[9,1] | 1[1] | 1[1] | 1[1] | 2[11] |
| *Pomoxis nigromaculatus* | Black Crappie | I | 1[2,3] | 1[1] | 1[1] | 2[50,66] | 2[64,66] | 2[24,1] | 2[1] | 1[1] | 1[1] | 1[1] | 1[1] | 2[11] |
| **Cottidae** | | | | | | | | | | | | | | |
| *Cottus aleuticus* | Coastrange Sculpin | N | 3[5,67] | 3[1] | 2[5,1] | 2[50] | 2[1] | 1[24,1] | 2[1] | 2[1] | 2[1] | 2[1] | 1[1] | 1[1] |
| *C. asper* | Prickly Sculpin | N | 2[5,7,52,67] | 2[1] | 2[5,16] | 2[50,66] | 2[1,66] | 1[24,1] | 2[1] | 2[1] | 2[1] | 2[1] | 1[1] | 1[1] |
| *C. bairdi* | Mottled Sculpin | N | 3[7,44] | 2[1,44] | 2[1,44] | 3[50,66] | 2[44,64,66] | 1[24,1] | 2[1,44] | 2[1,44] | 2[1,44] | 2[1,44] | 1[1] | 1[25,1] |
| *C. cognatus* | Slimy Sculpin | N | 3[1,67] | 2[1] | 2[1] | 2[50] | 2[1] | 1[24,1] | 2[1] | 2[1] | 2[1] | 2[1] | 1[1] | 1[1] |
| *C. confusus* | Shorthead Sculpin | N | 3[5,7,34,67] | 3[34,44] | 3[5,34,44] | 2[50] | 2[1,34] | 1[24,1] | 2[1,34] | 2[34,44] | 2[34] | 2[34,44] | 1[34,44] | 1[1,34] |
| *C. ricei* | Spoonhead Sculpin | N | 3[1] | 2[1] | 2[1] | 2[50] | 1[1] | 1[24,1] | 2[1] | 2[1] | 2[1] | 2[1] | 1[71] | 1[1] |
| *C. rhotheus* | Torrent Sculpin | N | 3[5,67] | 2[1] | 2[5,1] | 3[50,66] | 2[1] | 1[24,1] | 2[1] | 2[1] | 2[1] | 2[1] | 1[1] | 1[1] |
| *C. sp.* | Cultus Lake Sculpin | N | 3[27] | 1[27,44] | 1[27,44] | 2[50] | 2[27,44] | 1[24,1] | 2[27,44] | 2[27] | 1[27,44] | 1[27,44] | 1[27,44] | 1[27] |
| **Cyprinidae** | | | | | | | | | | | | | | |
| *Acrocheilus alutaceus* | Chiselmouth | N | 1[5,16,19,44,67] | 2[1,44] | 2[5,16,19,44] | 2[49,1,66] | 2[1,66] | 2[24,1] | 2[1,44] | 3[1] | 2[1] | 1[1,44] | 1[1,44] | 1[1] |
| *Carassius auratus* | Goldfish | I | 1[2,52] | 1[1] | 1[1] | 1[50,1,66] | 1[1,66] | 2[24,1] | 1[1] | 1[1] | 1[1] | 1[1] | 1[1] | 1[1] |
| *Couesius plumbeus* | Lake Chub | N | 1[7,16,52,67] | 2[1] | 2[1,16] | 2[49,50] | 1[64] | 2[24,1] | 2[1] | 3[1] | 2[1] | 1[1] | 1[1] | 2[20] |
| *Cyprinus carpio* | Carp | I | 1[3,45,52] | 1[1] | 1[1] | 1[50,51,66] | 1[1,64,66] | 3[24,1] | 1[1] | 1[9,1] | 1[1] | 1[1] | 1[1] | 2[11] |
| *Hybognathus hankinsoni* | Brassy Minnow | N | 2[1] | 3[1,44] | 3[1,44] | 2[49,50] | 1[1,44] | 1[24,1] | 2[1,44] | 1[1,44] | 2[1] | 1[1] | 1[1,44] | 1[1] |
| *Margariscus margarita* | Pearl Dace | N | 2[1,44] | 2[1,44] | 2[1,44] | 2[49,50] | 2[1,44] | 1[24,1] | 2[1,44] | 3[1] | 2[1] | 1[1] | 1[1,44] | 1[1] |
| *Mylocheilus caurinus* | Peamouth Chub | N | 2[7,16,67] | 1[1] | 1[1,16] | 2[49,50] | 2[1] | 2[24,1] | 2[1] | 3[1] | 2[1] | 1[1] | 1[1] | 1[1] |
| *Notropis atherinoides* | Emerald Shiner | N | 2[2,1] | 1[1,44] | 1[1,44] | 2[49,50] | 2[44,64] | 1[24,1] | 2[1,44] | 1[1,44] | 1[1,44] | 1[1,44] | 1[1,44] | 1[11] |
| *N. hudsonius* | Spottail Shiner | N | 2[1] | 1[1,44] | 1[1,44] | 2[49,50] | 2[44,64] | 1[24,1] | 2[1,44] | 3[1,44] | 1[1,44] | 1[1,44] | 1[1,44] | 1[1] |
| *Phoxinus eos* | Redbelly Dace | N | 1[1,44] | 2[1] | 2[1,44] | 2[49,50] | 2[44,64] | 1[24,1] | 2[1,44] | 1[1,44] | 1[1] | 1[1] | 1[1,44] | 1[1] |
| *Phoxinus neogaeus* | Finescale Dace | N | 1[1] | 2[1] | 2[1] | 2[49,50] | 2[1] | 1[24,1] | 2[1,44] | 1[1] | 1[1] | 1[1] | 1[1] | 1[1] |
| *Pimephales promelas* | Fathead Minnow | I | 1[2,4] | 2[1] | 2[1] | 1[49,50,4,66] | 1[1,64,65] | 1[24,1] | 2[1] | 1[1,10] | 1[1] | 1[1] | 1[1] | 2[1] |
| *Platygobio gracilis* | Flathead Chub | N | 2[1] | 2[1] | 1[1] | 2[49,50] | 1[1] | 2[24,1] | 2[1] | 3[1] | 2[1] | 1[1] | 1[1] | 1[1] |

| Family/ Species name | Common name | Origin[a] | T | HS | HR | O2 | ST | RS | FM | SM | MS | WC | SS | C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Ptychocheilus oregonensis* | Northern Pike-minnow | N | 1 5,7,52,67 | 1 1 | 1 5,16 | 2 49,50,66 | 2 1,66 | 2 24,1 | 2 1 | 3 1 | 2 1,57 | 1 1 | 1 1 | 1 1 |
| *Rhinichthys cataractae* | Longnose Dace | N | 2 5,7,67 | 2 1 | 2 5,16 | 2 49,50,66 | 2 1,64,66 | 1 24,1 | 2 1 | 3 1 | 2 1 | 2 1 | 1 1 | 1 1,25 |
| *Rhinichthys osculus* | Speckled Dace | N | 1 7,44,67 | 2 29,30,44 | 1 30,44 | 2 49,50,66 | 2 29,44,66 | 1 24,1 | 2 29,44 | 3 1 | 2 1,29 | 2 29,44 | 1 29,44 | 1 29 |
| *Rhinichthys falcatus* | Leopard Dace | N | 1 16,35,52,67 | 1 1,35 | 1 1,16,35 | 2 49,50,66 | 1 28,35,66 | 1 24,1 | 2 1,35 | 3 1 | 2 35 | 2 28 | 1 1,35 | 1 1,35 |
| *Rhinichthys sp.* | Umatilla Dace | N | 2 26,44 | 1 26 | 1 26,44 | 2 49,50 | 2 26,44 | 1 24,1 | 2 26,44 | 3 1 | 2 26 | 2 26,44 | 1 26,44 | 1 26 |
| *Rhinichthys sp.* | Nooksack Dace | N | 2 21,67 | 3 21,44 | 3 21,44 | 2 49,50,66 | 2 21,44,66 | 1 24,1 | 2 21,44 | 3 21,44 | 1 21,44 | 2 21 | 1 21,44 | 1 21,44 |
| *Richardsonius balteatus* | Redside Shiner | N | 1 5,7,52,67 | 2 5,16 | 2 1 | 2 49,50,66 | 2 1,66 | 1 24,1 | 2 1 | 3 1 | 2 1,57 | 1 1 | 1 1 | 1 1 |
| *Tinca tinca* | Tench | I | 1 45 | 1 1 | 1 1 | 1 50,1 | 1 1 | 3 24,1 | 1 1 | 1 1 | 1 1 | 1 1 | 1 1 | 1 1 |
| **Catastomatidae** | | | | | | | | | | | | | | |
| *Catostomus catostomus* | Longnose Sucker | N | 3 7,52,67 | 2 1 | 2 1,16 | 3 1 | 1 1 | 3 24,1 | 1 1 | 3 1 | 2 1 | 1 1 | 1 1 | 1 1 |
| *C. columbianus* | Bridgelip Sucker | N | 2 5,66 | 2 1 | 2 5,16,66 | 2 50,1 | 1 1 | 2 24,1 | 1 1,66 | 3 1 | 2 1 | 1 1,66 | 1 1,66 | 1 1 |
| *C. commersoni* | White Sucker | N | 2 3,4,45,52 | 2 1 | 2 1 | 2 49,50,4 | 1 1 | 3 24,1 | 1 1 | 3 1,9 | 2 1 | 1 1 | 1 1 | 1 25 |
| *C. macrocheilus* | Largescale Sucker | N | 2 5,52,67 | 2 1 | 1 16,66 | 2 1,65 | 1 1,66 | 3 24,1 | 1 1,66 | 3 1 | 2 1 | 1 1,65 | 1 1,65 | 1 1 |
| *C. platyrhynchus* | Mountain Sucker | N | 3 7,44,67 | 3 1,44 | 3 40,44 | 3 40,44,65 | 1 1,44,66 | 2 24,1 | 1 40,44 | 3 1 | 2 1,40 | 2 1,44 | 1 40,44 | 1 1 |
| *C. sp.* | Salish Sucker | N | 2 21,47 | 3 21,44,47 | 3 21 14,47 | 3 21,47 | 2 21 | 3 24,1 | 1 21,44,47 | 3 21,47 | 2 21 | 1 21 | 1 21,44 | 1 21,44 |
| **Esocidae** | | | | | | | | | | | | | | |
| *Esox lucius* | Northern Pike | N | 2 3,6,45 | 1 6,1 | 1 6,1 | 1 49,50,51 | 2 1,64 | 3 24,1 | 2 6 | 1 6,1 | 1 6,1 | 1 6,1 | 1 6,1 | 1 6,1 |
| **Gadidae** | | | | | | | | | | | | | | |
| *Lota lota* | Burbot | N | 3 6,7,67 | 1 6,16 | 1 6,16 | 3 49,50 | 2 1,6 | 3 24,1 | 2 6,1 | 3 6,1 | 2 6,1 | 2 6,1 | 2 6,1 | 2 6 |
| **Gasterosteidae** | | | | | | | | | | | | | | |
| *Culea inconstans* | Brook Stickleback | N | 2 1,64,66 | 2 1,55 | 2 1,55 | 2 49,50 | 2 55,64 | 1 24,1 | 2 1,55 | 1 1,55 | 2 1 | 1 1,55 | 1 1,55 | 2 1 |
| *Gasterosteus aculeatus* | Threespine Stickleback | N | 1 5,7,66,67 | 2 1,55 | 2 5,1,55 | 2 65,49 | 2 1,65 | 1 24,1 | 2 1,55 | 1 1,55 | 3 1 | 1 1,55 | 1 1,55 | 2 20 |
| *G. sp.* | Unarmoured Stickleback | N | 2 1,67 | 1 38 | 1 38 | 2 1 | 2 38 | 1 24,1 | 2 38,44 | 1 1 | 1 38 | 1 38,44 | 1 38,44 | 1 38 |
| *G. sp.* | Lake Sticklebacks | N | 2 1,44,67 | 1 44 | 1 44 | 2 1 | 2 44 | 1 24,1 | 2 44 | 1 44 | 1 44 | 1 44 | 1 44 | 1 44 |
| *G. sp.* | Giant Black Stickleback | N | 2 1,44,67 | 1 37,44 | 1 37,44 | 2 1 | 2 37,44 | 1 24,1 | 2 37,44 | 1 37 | 1 37 | 1 37,44 | 1 37,44 | 1 37 |
| *Pungitius pungitius* | Ninespine Stickleback | N | 2 1,44 | 2 1,44,55 | 2 1,55 | 2 1 | 1 1 | 1 24,1 | 2 1,44,55 | 1 1,55 | 2 1,55 | 1 1,55 | 1 1,44,55 | 2 1 |
| **Hiodontidae** | | N | | | | | | | | | | | | |
| *Hiodon alosoides* | Goldeye | N | 2 1 | 1 1 | 1 1 | 2 49,50 | 1 1 | 3 24,1 | 1 1 | 1 1 | 3 1 | 1 1 | 1 1 | 1 1 |
| **Ictaluridae** | | | | | | | | | | | | | | |
| *Ameiurus melas* | Black Bullhead | I | 1 4,52 | 1 1 | 1 1 | 2 50,4 | 1 1,64 | 2 24,1 | 1 1 | 1 9,1 | 1 1 | 1 1 | 1 1 | 1 1 |
| *A. nebulosus* | Brown Bullhead | I | 1 3 | 1 1 | 1 1 | 1 50,51 | 1 1,64 | 2 24,1 | 1 1 | 1 1 | 1 1 | 1 1 | 1 1 | 2 1 |
| **Osmeridae** | | | | | | | | | | | | | | |
| *Spirinchus thaleichthys* | Longfin Smelt | N | 3 7,67 | 1 1,36 | 1 1,36 | 3 1 | 2 1,36 | 1 24,1 | 2 1 | 3 1 | 3 1 | 1 1 | 2 1 | 1 1,36 |
| *S.sp.* | Pygmy Longfin Smelt | N | 3 7,67 | 1 1,36 | 1 44 | 3 1 | 2 1,44 | 1 24,1 | 2 44 | 3 1 | 2 1 | 1 44 | 2 44 | 1 1,36 |
| *Thaleichthys pacificus* | Eulachon | N | 3 7,67 | 1 1 | 1 1 | 3 1 | 2 1 | 1 24,1 | 1 1 | 3 1 | 3 1 | 1 1 | 1 1 | 1 1 |

| Family/ Species name | Common name | Origin* | T | HS | HR | O2 | ST | RS | FM | SM | MS | WC | SS | C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Percidae** | | | | | | | | | | | | | | |
| *Perca flavescens* | Yellow Perch | I | 2 2,3,45,52 | 1 1 | 1 1 | 1 49,50 | 2 6,64 | 2 24,1 | 2 1 | 1 1 | 1 1 | 1 1 | 1 1 | 1 11 |
| *Stizostedion vitreum* | Walleye | N | 2 2,3,6,45 | 1 6,1 | 1 6,1 | 2 49,51 | 1 1 | 3 24,1 | 1 6,1 | 3 6,1 | 2 6,1 | 1 6,1 | 1 6,1 | 2 11,6 |
| **Percopsidae** | | | | | | | | | | | | | | |
| *Percopsis omiscomaycus* | Troutperch | N | 3 2 | 2 1 | 1 1 | 2 1 | 1 1 | 1 24,1 | 2 1 | 3 1 | 2 1 | 1 1 | 1 1 | 1 1 |
| **Petromyzidae** | | | | | | | | | | | | | | |
| *Lampetra ayresi* | River Lamprey | N | 3 58,67 | 1 1,62 | 1 1,62 | 3 58,63,1 | 1 1,59 | 3 24,1 | 2 1,60 | 2 1,61 | 3 1,61 | 1 59,61,62 | 1 1,62 | 1 1,59 |
| *L. macrostoma* | Lake Lamprey | N | 3 22,58 | 1 22,44,46 | 1 22,44,46 | 3 44,58,63 | 1 22,44,46,58 | 3 22 | 2 22,44,46,60 | 2 22,46,61 | 1 22,44,46 | 1 22,46,59,61 | 1 22,46 | 1 22,46,59 |
| *L. richardsoni* | Brook Lamprey | N | 3 5,7,58,67 | 2 5,1 | 2 5,1 | 3 58,63,1 | 1 1,59 | 3 24,1 | 2 1,60 | 2 1,61 | 1 1,61 | 1 58,61 | 1 1,62 | 1 1,59 |
| *L. richardsoni marifuga* | Morrison Creek Lamprey | N | 3 33,58,67 | 3 1,33 | 3 1,33 | 3 58,63,1 | 1 33,59 | 3 24,1 | 2 33,60 | 2 33,61 | 1 1,33,61 | 1 33,59,61 | 1 1,33 | 1 33,59 |
| *L. tridentata* | Pacific Lamprey | N | 2 5,1,58,67 | 3 5,62 | 2 5,62 | 3 58,63,1 | 1 1,59 | 3 24,1 | 2 1,60 | 2 1,61 | 3 1,61 | 1 59,61,62 | 1 1,62 | 1 1,59 |
| **Salmonidae** | | | | | | | | | | | | | | |
| *Coregonus artedii* | Lake Cisco | N | 3 44,45,53 | 1 1,44 | 1 1,44 | 3 49,1 | 2 1,44 | 2 24,1 | 2 1,44 | 3 1,44 | 1 1,44 | 1 1,44 | 2 1,44 | 1 1 |
| *C. autumnalis* | Arctic Cisco | N | 3 1 | 1 1 | 1 1 | 3 49,1 | 2 1 | 3 24,1 | 2 1 | 3 1 | 2 1 | 1 1 | 2 1 | 1 1 |
| *C. clupeaformis* | Lake Whitefish | N | 3 6,1 | 1 6 | 1 6 | 3 49,1 | 2 64 | 2 24,1 | 2 6,1 | 3 6,1 | 1 6 | 1 6,1 | 2 6,1 | 2 6 |
| *C. nasus* | Broad Whitefish | N | 3 1,44 | 1 1,44 | 1 1,44 | 3 49,51 | 2 1 | 3 24,1 | 2 1,44 | 3 1 | 2 1 | 1 1,44 | 2 1,44 | 1 1 |
| *C. sardinella* | Least Cisco | N | 3 1,44 | 1 1,44 | 1 1,44 | 3 49,1 | 2 1 | 2 24,1 | 2 1,44 | 3 1,44 | 2 1,44 | 1 1,44 | 2 1,44 | 1 1 |
| *Oncorynchus clarki clarki* | Cutthroat Trout | N | 3 3,5,7,67 | 3 5,15,41 | 3 5,15,41 | 3 49,1,65 | 2 6,41,65 | 2 24,1 | 2 1,41 | 2 1,41 | 3 1,41 | 1 1,41 | 1 1,41 | 2 15 |
| *O. clarki lewisi* | Westslope Cutthroat Trout | N | 3 6 | 3 6,41 | 3 6,1 | 3 49,1,65 | 2 6,1,65 | 2 24,1 | 2 6,1 | 2 6,1 | 3 6,1 | 1 6,1 | 1 6,1,41 | 2 6,1 |
| *O. gorbuscha* | Pink Salmon | N | 3 2,3 | 2 1,41 | 1 1,41 | 3 49,50 | 2 1 | 1 24,1 | 1 1 | 2 1,14 | 2 1,14 | 1 1,41 | 2 1,14 | 1 1 |
| *O. keta* | Chum Salmon | N | 3 2,3 | 2 1,14 | 1 1,14 | 3 49,50 | 2 1,41 | 2 24,1 | 2 14,41 | 2 1,14 | 3 1,14,41 | 1 1,41 | 2 14,41 | 1 1 |
| *O. kisutch* | Coho Salmon | N | 3 2,5,7,67 | 3 5,14,15 | 3 5,14,15,41 | 3 49,50,51,66 | 2 6,65 | 2 24,1 | 2 12,14,41 | 2 1,14 | 3 1,14 | 1 1,14 | 2 14,41 | 2 41,42 |
| *O. mykiss* | Rainbow Trout | N | 3 2,3,45,67 | 3 5,6,15 | 2 5,6,15 | 3 49,50,51,66 | 2 6,41,64,66 | 2 24,1 | 2 12,1 | 2 6,1 | 2 6,41 | 1 6,1 | 1 6,1 | 2 6 |
| *O. mykiss* | Steelhead Trout | N | 3 2,7,45,67 | 3 1,41 | 1 1,41 | 3 49,50,51,66 | 2 6,41,64,66 | 2 24,1 | 2 1 | 2 1,41 | 3 1,41 | 1 1,41 | 1 1,41 | 2 1,43 |
| *O. nerka* | Sockeye Salmon | N | 3 2,7,67 | 2 14,41 | 1 14,41 | 3 49,50,51 | 2 6,1 | 2 24,1 | 2 14,41 | 2 14 | 3 1,14 | 1 1,41 | 2 1,14,41 | 1 1,14 |
| *O. nerka* | Kokanee | N | 3 6,1 | 2 6,1,41 | 1 6,41 | 3 49,1 | 2 6,1 | 2 24,1 | 2 6,41 | 2 6,1 | 2 6,1 | 1 6,1 | 2 6,1,41 | 2 6,1 |
| *O. tshawytscha* | Chinook Salmon | N | 3 2,7,67 | 2 14,15,41 | 2 14,15,15,41 | 3 49,50,51,66 | 2 6,65 | 3 24,1 | 2 12,14 | 2 1,14 | 3 1,14 | 1 1,41 | 2 1,14,41 | 2 7 |
| *Prosopium coulteri* | Pygmy Whitefish | N | 3 1 | 1 1 | 1 1 | 3 49,1 | 2 1 | 1 24,1 | 2 1 | 3 1 | 1 1 | 1 1 | 2 1 | 1 1 |
| *P. cylindraceum* | Round Whitefish | N | 3 2 | 1 1 | 1 1 | 3 49,1 | 2 1 | 3 24,1 | 2 1 | 3 1 | 1 1 | 1 1 | 2 1 | 1 1 |
| *P. sp.* | Giant Pygmy Whitefish | N | 3 31 | 1 31,44 | 1 31,44 | 3 49,1 | 2 31,44 | 2 24,1 | 2 31 | 3 1 | 1 31,44 | 1 31,44 | 2 31,44 | 1 31 |
| *P. williamsoni* | Mountain Whitefish | N | 3 3,6 | 2 6,15 | 2 6,15 | 3 49,65 | 2 1 | 2 24,1 | 2 6,1,44 | 3 6,1 | 3 6,32 | 1 6,1 | 2 6,1 | 2 6 |
| *Salmo salar* | Atlantic Salmon | I | 3 2,70 | 2 1,41 | 2 1,41 | 3 49,50,51 | 2 6,1 | 2 24,1 | 2 1,41 | 2 1,41 | 3 1,70 | 1 1,41 | 2 1,41 | 1 7 |
| *Salmo trutta* | Brown Trout | I | 3 2,3 | 3 1,41 | 2 1,41 | 3 49,50,51 | 2 6,64 | 2 24,1 | 2 1 | 2 1,41 | 2 1,41 | 1 1,41 | 2 6, 17,44 | 2 1,25 |
| *Salvelinus confluentus* | Bull Trout | N | 3 5,7,17,44,67 | 3 5,17,41,44 | 2 5,17,41,44 | 3 49,1 | 2 6,1 | 3 17 | 2 17,41,44 | 2 6,17 | 3 6,17,41,44 | 1 6,17 | 2 1,13,41 | 2 17,44 |
| *Salvelinus fontinalis* | Brook Trout | I | 3 2,3,45 | 3 6,1 | 2 6,1 | 3 49,50,51 | 2 6,64 | 2 24,1 | 2 1,41 | 2 6,1 | 2 6,1,41 | 1 1,41 | 2 1,25 | 2 25 |

| Family/ Species name | Common name | Origin* | T | HS | HR | O2 | ST | RS | FM | SM | MS | WC | SS | C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Salvelinus malma* | Dolly Varden | N | 3[7,1] | 3[41,42] | 3[41,42] | 3[40,1] | 2[6,1] | 3[24,1] | 2[1,41] | 2[1] | 3[1,41] | 1[1,41] | 2[1,41] | 2[42] |
| *Salvelinus namaycush* | Lake Trout | N | 3[2,6] | 1[6,41] | 1[6,41] | 3[49,1] | 2[6,1] | 3[24,1] | 2[1,41] | 3[6,41] | 1[6,41] | 1[6,1] | 2[6,1] | 2[6] |
| *Stenodus leuchichthys* | Inconnu | N | 3[1] | 1[1,66] | 1[1,66] | 3[1] | 2[1] | 3[24,1] | 2[1] | 3[1] | 3[1,66] | 1[1] | 2[1,66] | 1[1] |
| *Thymallus arcticus* | Arctic Grayling | N | 3[2,6,44] | 3[6,44] | 2[6,44] | 3[6,1] | 2[6,44] | 2[24,1] | 2[6, 12,1,44] | 3[6,44] | 3[6,44] | 1[6,44] | 1[1,44] | 1[6] |

* N = native species, I = introduced species
T = hyperthermia tolerance
HS = Spawning habitat
HR = rearing habitat
O2 = hpoxia tolerance
ST = siltation tolerance
RS = reproductive strategy
FM = feeding mode
SM =spawning mode
MS = migration strategy
WC = water column/substrate use
SS = spawning season
C = use of LWD cover

## References for scoring matrix:


1) Scott, W.B., and E.J. Crossman. 1973. Freshwater Fishes of Canada. Fisheries Research Board of Canada Bulletin 184. Ottawa, Ontario.
2) Coutant, C.C. 1977. Compilation of temperature preference data. Journal of Fisheries Research Board Canada 34: 739-745.
3) Eaton, J.G., J.. McCormick, B.E. Goodno, D.G. O'Brien, H.G. Stefany, M.Hondzo, and R.M. Scheller. 1995. A field information-based system for estimating fish temperature tolerances. Fisheries 20(4): 10-18.
4) Smale, M.A., and C.F. Rabeni. 1995. Hypoxia and hyperthermia tolerances of headwater stream fishes. Transactions of the American Fisheries Society 124: 698-710.
5) Bond, C.E., and R.M. Hughes. 1988. Habitat use of Twenty-five common species of Oregon freshwater fishes. Northwest Science 62(5): 223-232.
6) Ford, B.S., P.S. Higgins, A.F. Lewis, K.L. Cooper, T.A. Watson, G.L. Ennis, and R.L. Sweeting. 1992. Literature reviews of the life history, habitat requirements and mitigation/compensation strategies for 13 species of sport fish in the Peace and Columbia River drainages of British Columbia.Canadian Department of Fisheries and Oceans Report, Vancouver, B.C., Canada.
7) Hicks, M. 1999. Evaluating standards for protecting aquatic life in Washington's surface water quality standards. Draft Discussion Paper. Water Quality Program. Watershed Management Section. Olympia, Washington.
8) Newcombe, C.P., and D.D. McDonald. 1991. Effects of suspended sediment on aquatic ecosystems. North American Journal of Fisheries Management 11: 72-82.
9) Berkman, H.E., and C.F. Rabeni. 1987. Effect of siltation on stream fish communities. Environmental biology of fishes 18(4): 285-294.
10) Johnston, C.E. 1999. The relationship of spawning mode to conservation of North American minnows (Cyprinidae). Environmental Biology of Fishes 55: 21-30.
11) Lehtinen, R.M., N.D. Mundahl, and J.C. Madejczyk. 1997. Autumn use of woody snags by fishes in backwater and channel border habitats of a large river. Environmental Biology of Fishes 49: 7-19.
12) Lloyd, D.S., J.P. Koening, and J.D. LaPierre. 1987. Effects of turbidity in fresh waters of Alaska. North American Journal of Fisheries Management 7: 18-33.
13) Erman, D.C., E.D. Andrews, and M. Yoder-Williams. 1988. Effects of winter floods on fishes in the Sierra Nevada. Canadian Journal of Fisheries and Aquatic Sciences 45: 2195-2200.
14) Groot C., and L. Margolis (eds.). 1991. Pacific salmon life histories. UBC Press. University of British Columbia, Vancouver, BC.
15) Rosenfeld, J., M. Porter, and E.Parkinson. 2000. Habitat factors affecting the abundance and distribution of juvenile cutthroat trout (Oncorhynchus clarki) and coho salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 57:766-774.
16) Porter, M., J. Rosenfeld, and E. Parkinson. 1998. Macrohabitat use and predictive models of fish distribution in the Blackwater drainage. Fisheries Province of British Columbia,

Ministry of Fisheries, Fisheries Management Report No. 106.
17) McPhail, J.D., and J.S. Baxter. 1996. A review of bull trout (Salvelinus confluentus) life-history and habitat use in relation to compensation and improvement opportunities. Province of British Columbia, Ministry of Evironment, Lands and Parks, Fisheries Management Report No. 104.
18) Murray, C.B., and M.L. Rosenau. 1989. Rearing of juvenile chinook salmon in nonnatal tributaries of the lower Fraser River, British Columbia. Transactions of the American Fisheries Society 118: 284-289.
19) Rosenfeld, J., B. Wicks, M. Porter, P. Van Dishoek. 1998. Habitat use by chiselmouth (Acrocheilus alutaceus) in the Blackwater River. Province of British Columbia, Ministry of Fisheries, Project Report No.RD76.
20) Rosenfeld, J. BC Fisheries. Fisheries Research Section. University of British Columbia. Personal communication.
21) Pearson, M. 1998. A review of the distribution, status, and biology of the endangered Salish Sucker (Catastomus sp.) and Nooksack Dace (Rhinicthys sp.). Provincre of British Columbia, Ministry of Fisheries, Fisheries Technical Circular No. 101.
22) Beamish, R.J. 1982. Lampetra macrostoma, a new species of freshwater lamprey from the west coast of Canada. Canadian Journal of Fisheries and Aquatic Sciences 39: 736-747.
23) Hart, N.B. 1973. Pacific fishes of Canada. Fisheries Research Board of Canada Bulletin 180. Ottawa, Ontario.
24) Anonymous. FishBase 99. A global information system on fishes. http://www.fishbase.org/
25) Cunjak, R.A. 1996. Winter habitat of selected stream fishes and potential impacts from land-use activity. Canadian Journal of Fisheries and Aquatic Sciences 53(Suppl. 1): 267-282.
26) Hughes, G.W., and A.E. Peden. 1989. Status of the Umatilla Dace, *Rhinicthys umatilla*, in Canada. Pages 193-200. *In* R.R. Campbell (ed.). Rare and endangered fishes and marine mammals of Canada: Status Reports: V. The Canadian Field Naturalist 103(2): 147-220.
27) Ikusemiju, K. 1967. The life history and ecology of *Cottus sp.* In Lake Washington. Masters thesis, College of Fisheries, University of Washington, Seattle, WA, U.S.A.
28) Gee, J.H., and T.G. Northcote. 1963. Comparative ecology of two sympatric species of dace (Rhinichthys) in the Fraser River System, British Columbia. Journal of the Fisheries Research Board Canada 20(1): 105-118.
29) Peden, A.E., and G.W. Hughes. 1981. Life history relevant to the Canadian Status of the speckled dace *(Rhinichthys osculus)*. Syesis 14: 21-31.
30) Peden, A.E., and G.W. Hughes. 1984. Status of the Speckled Dace, *Rhinicthys osculus*, in Canada. Canadian Field Naturalist 98(1): 98-103.
31) Rankin, L. 1999. Phylogenetic and ecological relationships between "Giant" Pygmy Whitefish *(Prosopium spp.)*and Pygmy Whitefish *(Prosopium coulteri) in North-Central* British Columbia. Masters thesis, University of Northern British Columbia, Prince George, B.C., Canada.
32) Northcote, T.G. 1997. Potomodromy in salmonida - living and moving in the fast lane. North American Journal of Fisheries Management 17: 1029-1045.
33) Beamish, R.J. 1987. Evidence that parasitic and nonparasitic life history types are produced by one population of lamprey. Canadian Journal of Fisheries and Aquatic Sciences 44: 1779-1782.
34) Peden, A.E., and G.W. Hughes. Status of shorthead sculpin, *Cottus confusus*, in the Flathead River, British Columbia. Canadian Field-Naturalist 98(1): 127-133.
35) Peden, A.E. 1990. Status of the Leopard Dace, *Rhinichthys falcatus*, in Canada. Canadian Field-Naturalist 105(2): 179-188.
36) Dryfoos, R.L. 1965. The life history and ecology of the longfin smelt in Lake Washington. Phd thesis, University of Washington, Seattle, WA, U.S.A.
37) Moodie, G.E. 1984. Status of the Giant (Mayer Lake) Stickleback, *Gasterosteus sp.*, on the Queen Charlotte Islands, British Columbia. Canadian Field-Naturalist 98(1): 115-119.
38) Reimchen, T.E. 1984. Status of the unarmoured and spine-deficient populations (Charlotte Unarmoured Stickleback) of threespine stickleback, Gastersteus sp., on the Queen Charlotte Islands, British Columbia. Canadian Field-Naturalist 98(1): 120-126.
39) Bodaly, R.A., J.W. Clayton, and C.C. Lindsey. 1988. Status of the Squanga whitefish, Coregonus sp., in the Yukon Territory, Canada. Canadian Field-Naturalist 102(1): 114-125.
40) Hauser, W.J. 1969. Life history of the mountain sucker, *Catastomus platyrhynchus*, in Montana. Transactions of the American Fisheries Society 98: 209-215.

41) Meehan, W.R., and T.C.. Bjornn. 1991. Salmonid distributions and life histories. Pages 47-82 *in* W.R. Meehan (editor). Influences of forest and rangeland management on salmonid fishes and their habitats. Special publication 19, American Fisheries Society, Bethesda, MD.
42) Elliot, S.T. 1986. Reduction of a Dolly Varden population and macrobenthos after removal of logging debris. Transactions of the American Fisheries Society 115: 392-347.
43) Hartman, G.F. 1965. The role of behaviour in the ecology and interaction of underyearling coho salmon (*Oncorhynchus kisutch*) and (steelhead trout (Salmo gairdneri). Journal of the Fisheries Research Board of Canada 20: 1035-1080.
44) Cannings, S.G., and J.Ptolemy. 1998. Rare freshwater fish of British Columbia. Province of British Columbia. B.C. Ministry of Environment, Lands and Parks. Conservation Data Centre. Victoria, B.C.
45) Hokanson, K.E.F. 1977. Temperature requirements of some percids and adaptations to the seasonal temperature cycle. Journal of the Fisheries Research Board of Canada 34: 1524-1550.
46) Beamish, R.J. 1987. Status of the Lake Lamprey, *Lampetra macrostoma*, in Canada. Canadian Field-Naturalist 101(2): 186-189.
47) McPhail, J.D. 1987. Status of the Salish Sucker, *Catastomus sp.*, in Canada. Canadian Field-Naturalist 101(2): 231-236.
48) Lane, E.D. 1991. Status of the White Sturgeon, *Acipenser transmontanus*, in Canada. Canadian Field-Naturalist 105(2): 161-168.
49) Barton, B.A., and B.R. Taylor. 1994. Dissolved oxygen requirements for fish of the Peace, Athabasca and Slave River basins. Northern Rivers Basins Study Project Report No.29. Edmonton, Alberta.
50) Doudoroff, P., and D.L. Shumway. 1970. Dissolved oxygen requirements of freshwater fishes. Food and Agriculture Organization of the United Nations. FOA Technical Report No. 86.
51) Davis, J.C. 1975. Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review. Journal of the Fisheries Research Board of Canada 32: 2295-2332.
52) Black, E.C. 1953. Upper lethal temperatures of some British Columbia Freshwater fishes. Journal of the Fisheries Research Board of Canada 10(4): 196-210.
53) Edsall, T.A., and P.J. Colby. 1970. Temperature tolerance of young-of-the-year cisco, *Coregonus artedii*.Transactions of the American Fisheries Society 99(3): 526-531.
54) Winn, H.E. 1960. Biology of the brook stickleback *Eucalia inconstans* (Kirtland). American Midland Naturalist 63(2): 424-438.
55) Wooton, R.J. 1976. The biology of the sticklebacks. Academic Press Inc. (London) Ltd. London, UK.
56) Wooton, R.J. 1984. A functional biology of sticklebacks. Croom Helm Ltd. London & Sydney.
57) Smith, R.J.F. 1991. Social behaviour, homing and migration. *In* Winfield, I.J. and J.S. Nelson (eds.). Cyprinid Fishes: Systematics, Biology and Exploitation Chapman and Hall (London).
58) Potter, I.C. 1980. The Petromyzoniformes with particular reference to paired species. Canadian Journal of Fisheries and Aquatic Sciences 37: 1595-1615.
59) Potter, I.C. 1980. Ecology of larval and metamorphosing lampreys. Canadian Journal of Fisheries and Aquatic Sciences 37: 1641-1657.
60) Moore, J.W. 1980. Feeding of larval lamprey. Canadian Journal of Fisheries and Aquatic Sciences 37: 1658-1664.
61) Larsen, L.O. 1980. Physiology of adult lampreys, with special regard to natural starvation, reproduction and death after spawning. Canadian Journal of Fisheries and Aquatic Sciences 37: 1762-1779.
62) Beamish, R.J. Adult biology of the river lamprey (*Lampetra ayresi*) and the Pacific lamprey (*Lampetra tridentata*) from the Pacific coast of Canada. Canadian Journal of Fisheries and Aquatic Sciences 37: 1906-1923.
63) Lewis, S.V. 1980. Respiration of lampreys. Canadian Journal of Fisheries and Aquatic Sciences 37: 1711-1722.
64) Poff, N.L., and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76(2): 606-627.
65) Fuller, W.A. 1955. The Inconnu (Stenodus leucichthys) in Great Slave Lake and adjoining waters. Journal of the Fisheries Research Board of Canada 12(5): 768-780.
66) Hughes, R.M., P.K. Kaufman, A.T. Herlihy, T.M Kincaid, L. Reynolds, and D.P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. Canadian Journal of Fisheries and Aquatic Sciences 55: 1618-1631.

66) Dauble, D.D. 1986. Life history and ecology of the largescale sucker (*Catastomus macrocheilus*) in the Columbia River. The American Midland Naturalist 116(2): 356-367.
67) Li, H.W., C.B. Schreck, C.E. Bond, and E. Rexstad. 1987. Factors influencing change in fish assemblages of Pacific Northwest streams. *In* W.J. Mathews and D.C. Heins. (editors). Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman. Oklahoma, USA.
68) Dauble. D.D. 1980. Life history of the bridgelip sucker in the Central Columbia River. Transactions of the American Fisheries Society 109: 92-98.
70) Mills, D. 1989. Ecology and management of Atlantic salmon. Chapman and Hall, London, UK.